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ECOLOGICAL ASSESSMENTS FOR WASTEWATER  
MANAGEMENT IN SOUTHEASTERN MICHIGAN

By

10 Thomas G. Bahr  
Project Coordinator

9 Final report on Phase 2,

\*Technical Report No. 29

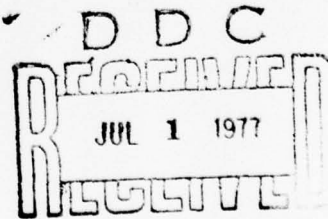
3 Institute of Water Research  
1 Michigan State University  
2 East Lansing, Michigan 48823

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## PREFACE

In the spring of 1971 the Institute of Water Research at Michigan State University completed the preliminary assessments of ecological impacts of wastewater management alternatives proposed by the Corps of Engineers for the Southeastern portion of Michigan. Following this early feasibility stage the Corps decided to pursue the refinement of their engineering plans and conduct a more in-depth analysis of environmental impacts that might result from implementing these plans. The Institute of Water Research formally entered into contract on May 8, 1972 to undertake these assessments and, with the assistance of approximately fifteen Michigan State University scientists, proceeded working to meet an October 1, 1972 completion date. Within this period we sought to meet two general objectives: (1) to characterize the present ecological status of Southeast Michigan Study area using existing stores of environmental data, and (2) to develop estimates of environmental change based on the initial characterization and the proposed environmental alternation schemes.

Although research involving the synthesis of existing data often generates considerable insight into problems that have received little attention in the past, this approach is generally not consistant with strict scientific methodology. Thus, it is not surprising that highly qualified ecologists and environmental scientists hesitate to become involved in such data synthesizing exercises. Many of the project participants on this study shared this hesitation, but nevertheless agreed to involve themselves in this task. This is a credit to both them and their profession. All participants are established scientists with in-depth knowledge of their subject area and each have a history of active involvement in basic and applied research directly relating to the objectives of this study.

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The conclusions advanced in this report represent the best estimates that could be generated in the short time span of this study. Due to the nature of the input from the project participants it would be difficult to designate specific sections of this report with individual scientists. Rather, findings were blended into the text of the report in order to achieve a more orderly presentation of material.

#### Project Participants

Thomas G. Bahr -- Assistant Director, Institute of Water Research,  
Project Coordinator (Limnology and Pollution Biology)

Robert C. Ball -- Director, Institute of Water Research,  
(Limnology and Pollution Biology)

James O. Corlett -- Research Associate, Institute of Water Research,  
(Project Engineer - Mass Balance Studies)

Frank M. D'Itri -- Assistant Professor, Institute of Water Research,  
(Project Chemist - Heavy Metals)

Richard A. Cole -- Research Associate, Institute of Water Research  
(Lake Erie Limnology)

\*Marvin E. Stephenson -- Associate Professor, Institute of Water  
Research, (Environmental Engineering)

Robert K. Johnson -- Research Assistant, Institute of Water Research,  
(Data Acquisition and Processing)

Boyd G. Ellis -- Professor, Department of Crop and Soil Sciences,  
(Consultant on Soil Chemistry - Phosphorus)

Arthur. R. Wolcott -- Professor, Department of Crop and Soil  
Sciences, (Consultant on Soil Nitrogen Dynamics)

Bernard D. Knezek -- Associate Professor, Department of Crop and  
Soil Sciences (Consultant on Soil Chemistry - Heavy Metals)

A. Earl Erickson -- Professor, Department of Crop and Soil Sciences,  
(Consultant on Soil Physics)

Niles R. Kevern -- Professor and Chairman, Department of Fisheries  
and Wildlife, (Consultant on Stream Ecology)

Howard E. Johnson -- Associated Professor, Department of Fisheries  
and Wildlife, (Consultant on Pesticides and Toxic Materials)  
Clarence D. McNabb -- Professor, Department of Fisheries and  
Wildlife, (Consultant on Aquatic Botany)  
Walter H. Conley -- Assistant Professor, Department of Fisheries  
and Wildlife, (Consultant on Terrestrial Ecology)  
Harold H. Prince -- Associate Professor, Department of Fisheries  
and Wildlife, (Consultant on Waterfowl Ecology)

\*Currently on leave of absence - present address: The Rockefeller  
Foundation, New York, New York.

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## SECTION I - GENERAL INTRODUCTION

### Purpose and Scope

This report was prepared in accordance with contractual obligations between the U.S. Army Corps of Engineers, Detroit District and The Institute of Water Research, Michigan State University, Contract Number, DACW 35-72-C-0031.

The general purpose of this study was to describe the ecological consequences of a variety of wastewater management alternatives proposed by the Corps of Engineers. These alternatives included advanced biological waste treatment, physical-chemical treatment, terrestrial disposal of waste and various combinations thereof.

General ecological considerations of these broad alternatives were discussed in an earlier report to the Corps of Engineers entitled "Alternatives for Managing Wastewater for Southeastern Michigan," Appendix F, July, 1971. Of necessity this early report considered only the more superficial ecological consequences of a given management strategy and, because of time constraints, available data could not be rigorously synthesized. In this study we probed significantly deeper into available literature and achieved a degree of data synthesis that enabled us to significantly sharpen our predictions of ecological consequences of waste management alternatives.

Our study is subdivided into two broad phases. The first phase was to establish a workable assessment methodology and collect necessary preliminary data. Included here was the development of a priority listing of data vectors of major ecological significance. The second phase of the study focused on the final assessment of management strategies selected by the Corps of Engineers. The information enclosed in this report represents the results of both phases of the study.

### Assessment Methodology

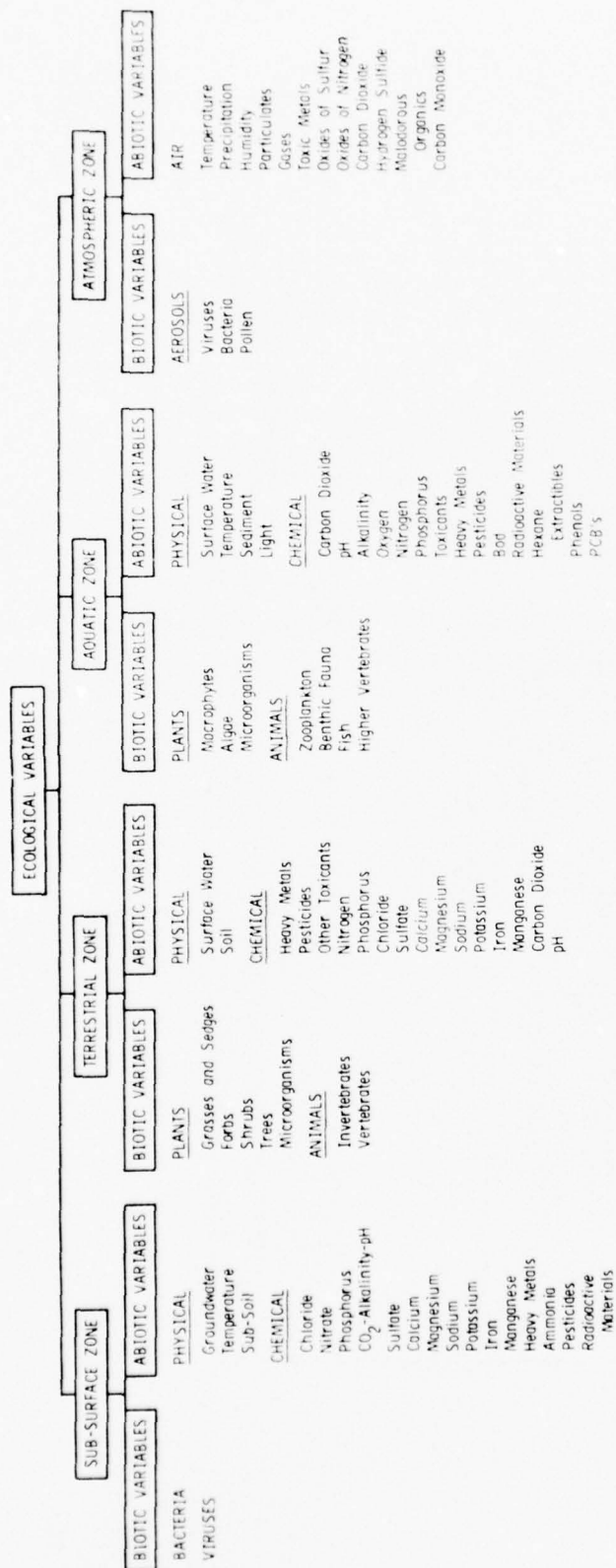
The only significant change from the Feasibility Study in terms of our assessment approach was to incorporate two additional components or ecological variables in our evaluation framework: (1) the assessment of depletion of potentially limiting resource reserves and (2) the assessment of impacts of energy utilization for a given wastewater management alternative. These two vectors represent ecological items that may appear somewhat insignificant on a local scale but they assume much greater relevance on a global scale. We have attempted to incorporate these items into our listing of ecological variables but they did not easily lend themselves to such compartmentalization. We thus consider them as a central thread running through all assessment phases. Figure 1 is the list of ecological variables just mentioned. This list is essentially unchanged from our earlier one from the Feasibility Study. We did, however, devote more energies in assessing certain components of Figure 1 than we did in the Feasibility Study. Our effort was roughly distributed as follows: (1) Subsurface Zone - <5%, (2) Terrestrial Zone - 15%, (3) Aquatic Zone - >75%, Atmospheric Zone - 5%.

The rationale behind this division of labor was based on the amount of time needed to synthesize data from the various areas, the ecological significance of the area from the standpoint of potential damage under a wastewater management scheme, and the results of our Feasibility Study. Within the aquatic zone we concentrated on the development of mass balances for phosphorus, nitrogen and carbon.

### Evaluation Framework

The means by which we arrived at the ecological impact of a particular waste management scheme is conceptually shown in Figure 2. The information and materials flow in Figure 2 starts in the upper left hand corner where biotic and abiotic components of the environment (food, minerals, etc.) are harvested and used by man as





SUBDIVISION OF MAJOR ECOLOGICAL VARIABLES

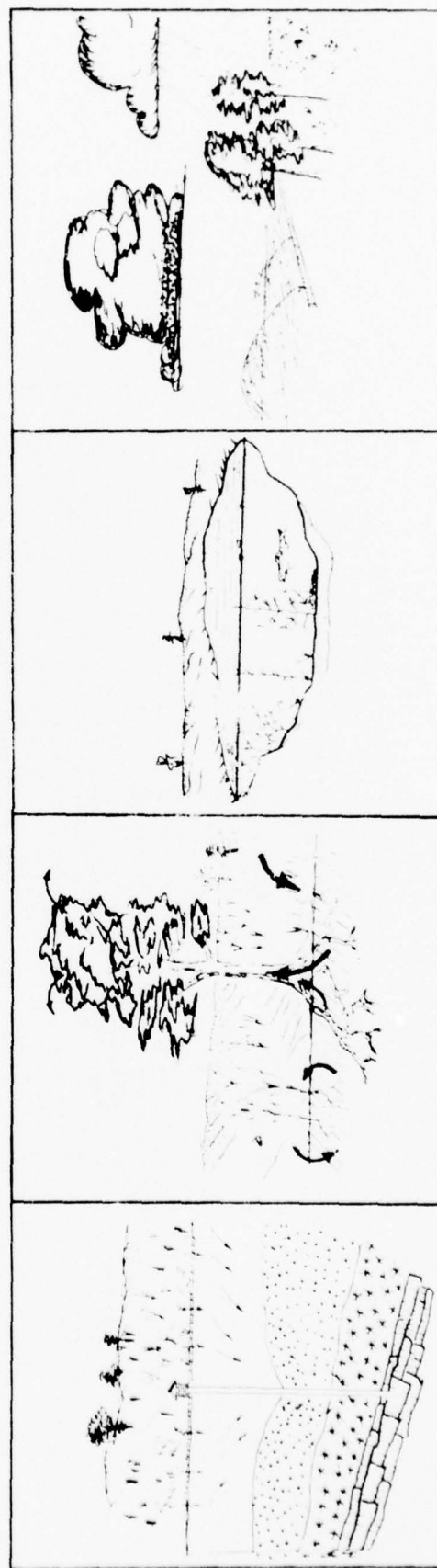


Figure 1. Ecological variables most likely to be affected by a wastewater management program.

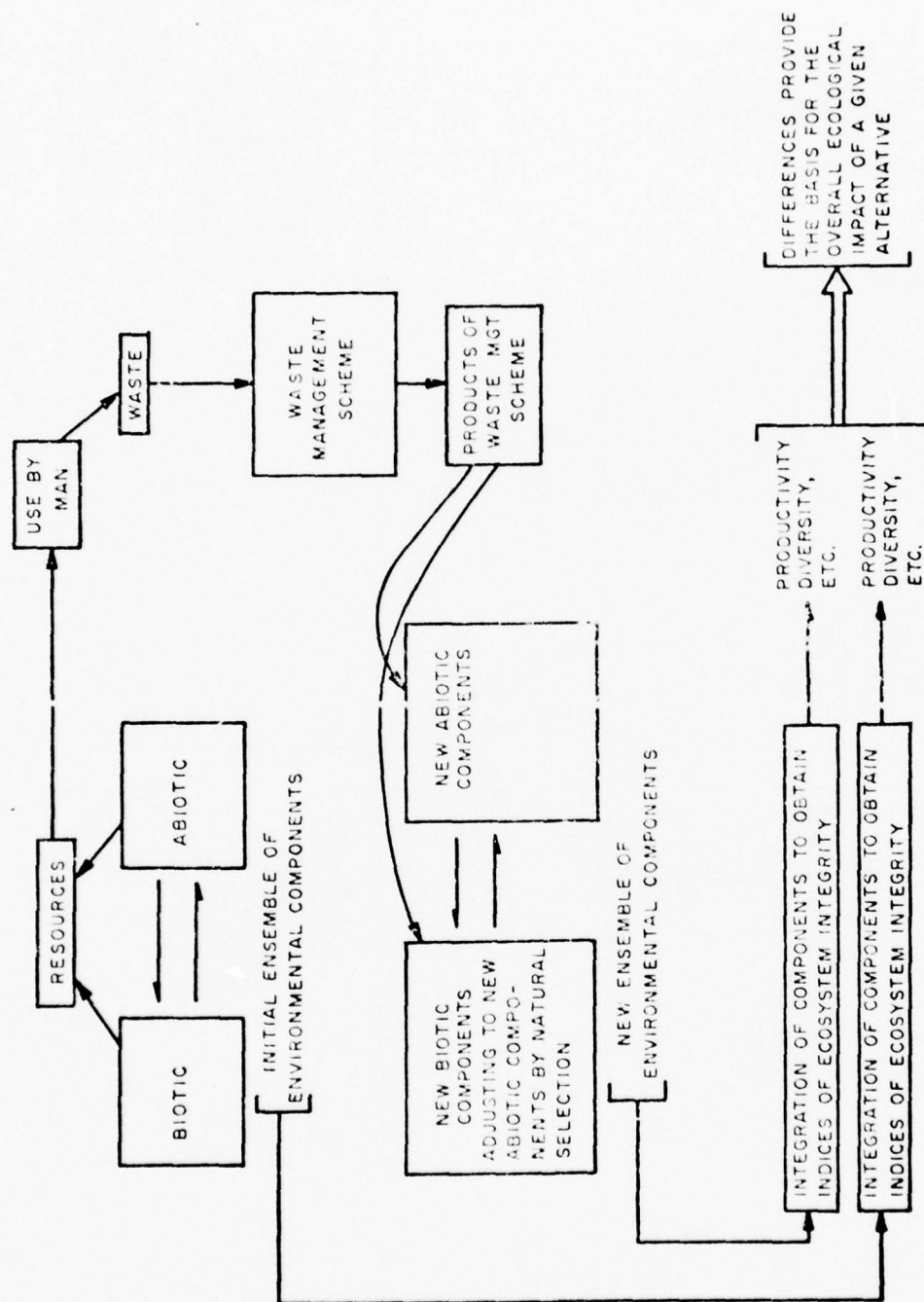


Figure 2. Flow diagram illustrating evaluation framework.

necessary resources. Waste products of these resources (municipal and industrial waste) are then fed into one of several waste management alternatives, are processed, and then returned to the environment as biproducts of the process. The majority of the biproducts are abiotic in nature (chemicals) although a few biotic components may also be released (bacteria and other microorganisms). There is established a new ensemble of environmental components (primarily abiotic in nature) resulting from the additional material input from the wastewater management scheme. The biotic components of the system will now adjust to the new abiotic nature of the environment and approach a new steady state condition. By measuring conditions and processes as they now exist and comparing them with anticipated environmental conditions we arrive at differences that form the basis of the environmental impact.

This approach proved to be very sound in earlier work and on its reevaluation we saw no reason to change it. We did, however, have the opportunity to greatly refine and quantify our integration scheme and more fully describe indices of ecosystem integrity. And as indicated earlier, we took a much closer look at the materials flux from "resources" to "use by man" as shown on the top of Figure 2. The bidirectional arrows connecting biotic and abiotic components in this figure represent the close interaction between these vectors. A more accurate representation of this interaction can be illustrated by considering the abiotic components as state variables that essentially dictate the nature of the biotic variable (solar energy, temperature, chemical concentration, etc.) that drive biotic components in the direction of a steady state condition. Changing the state variables will drive the biotic components toward a new steady state. This interaction is shown in the conceptual diagram of Figure 3 and is discussed more fully in Section IV of this report. Of importance here is the relative biomass and species composition of the primary producers, consumers, and decomposers. Changing a state variable, such as reducing

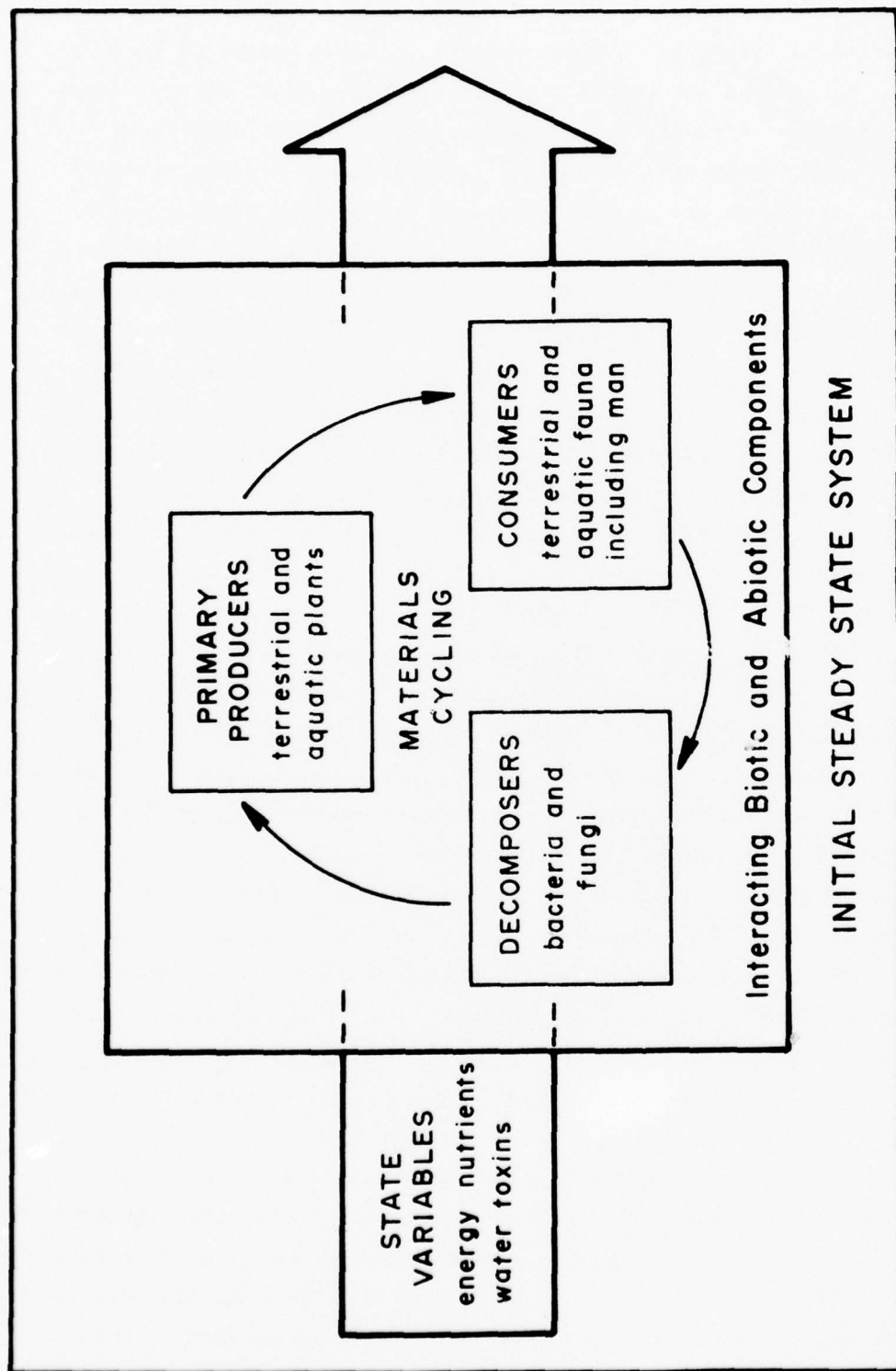


Figure 3. Interaction of Biotic and Abiotic Components.

phosphorus concentration in water below a threshold, will create a new biotic steady state that might be reflected in a reduction in the abundance of blue-green algae or increases in the abundance of oxygen intolerant fish species. This approach has a sound foundation in ecological theory and we exploited its use in our assessments.

## SECTION II - BASE CONDITIONS, BIOLOGICAL STRUCTURE

### Study Area

The impact of regionalized wastewater management for southeastern Michigan is certain to result in environmental change over geographical limits which greatly exceed the study area. Consequently, for the purposes of our assessment and evaluation it is necessary to define, at least approximately, an impact zone. Previous work indicates that this zone primarily includes the western sector of Lake Erie as well as those land masses contiguous to the study area which may receive translocated materials, e.g. wind-borne aerosols, etc. These areas could be considered the primary impact zone since the activities and byproducts of wastewater management schemes have a direct effect on the biotic and abiotic nature of the area. Secondary impact zones, however, will exist and these tend to be considerably more far reaching. Two such zones would include the central and eastern basins of Lake Erie. An even more remote example might be a strip mining area in Pennsylvania. Here, excessive fossil fuel power requirements, created by a particular wastewater management alternative for southeastern Michigan, would result in a corresponding increase in the demand for coal. This fuel supply may come from a strip mining region in Pennsylvania and thereby create a secondary impact on the terrestrial and aquatic systems of another state. Secondary impacts of this nature will be assessed and the affected areas identified.

For the purposes of analysis in this study we have designated the aquatic system as the primary impact zone and have subdivided it into nine regions. These are shown in Figure 4. The connecting waterway systems are comprised by the Niagara River (1), Maumee River (2), Detroit River (4), Southeastern Michigan Tributary Streams (8), St. Clair River (6), and the Southwestern Ontario (Canada) Tributary Streams (9). Also included is the Great Lakes system consisting of western Lake Erie (3), Lake St. Clair (5), and



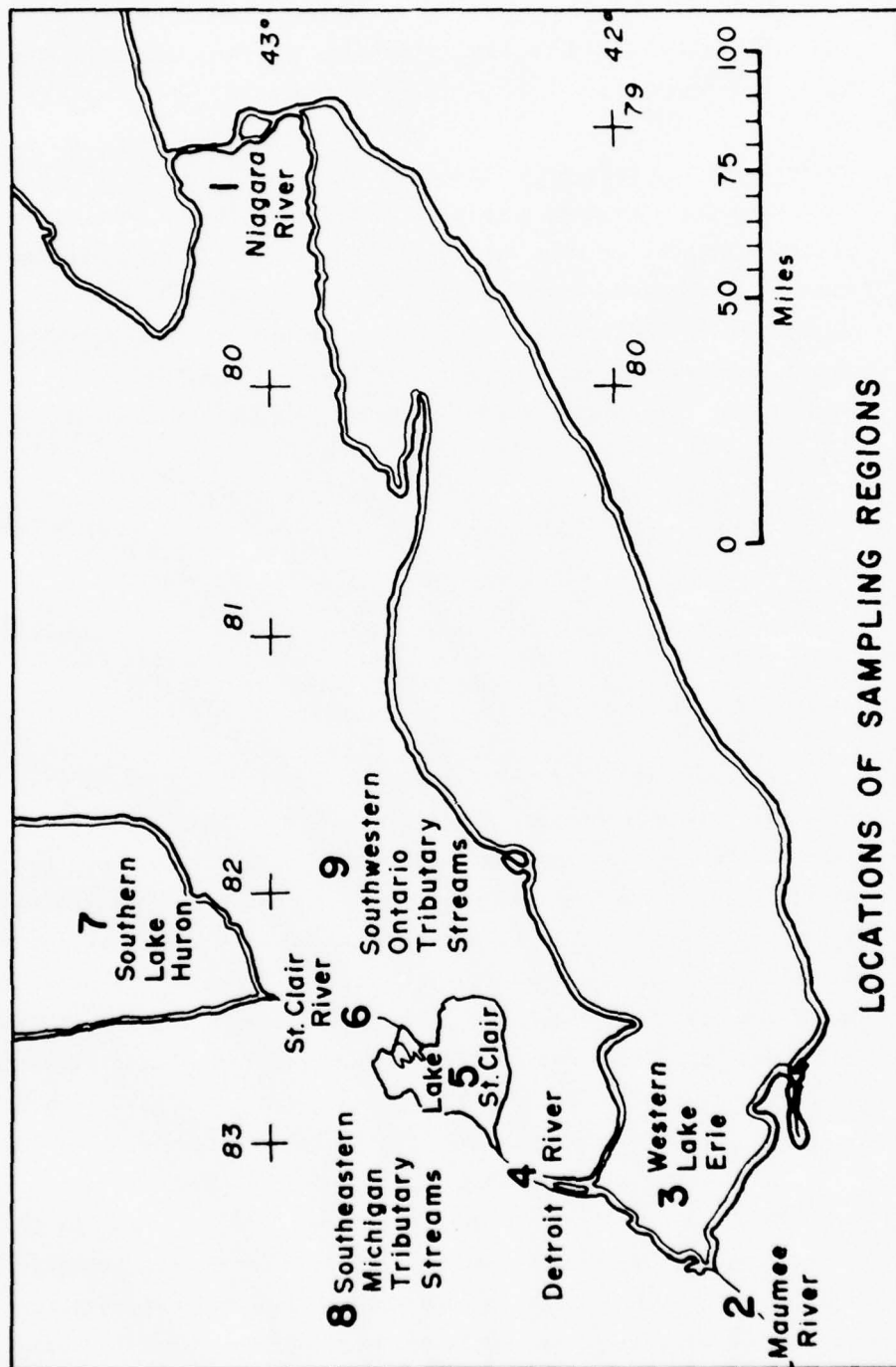


Figure 4.

southern Lake Huron (7). Within each of the regions listed above we have described, in detail, existing sampling points and have assigned a code number for each sampling station. The codes and descriptions can be found in Table 28.

#### Overview of the Lake Erie Problem

Lake Erie, by many popular accounts is dying or even dead. The negative aspects of Lake Erie have been forcefully presented by over eager nonprofessional activists, particularly in the news media, and in many quarters its demise is a foregone conclusion. Not surprisingly, even professionals have been influenced by rumors that the condition of Lake Erie approaches that of an open sewer.

*"As large a body of water as Lake Erie has already been overwhelmed by pollutants and has in effect, died. In its natural state, Lake Erie was a balanced system in which water plants, microorganisms, and a great variety of swimming creatures lived together in an intricate harmony. But today most of Lake Erie is dead. Sewage, industrial wastes, and the runoff from heavily fertilized farmlands have loaded the waters of the lake with so much excess phosphate and nitrate as to jar the biology of the lake permanently out of balance. The fish are all but gone."* (Commoner, 1966).

Certainly there has been enough research conducted on Lake Erie (more than any other of the worlds Great Lakes) to dispell much of the superstition that has evolved around the lake. However, the research that has been conducted has been characterized by isolated observation and experimentation with little regard for overall integration of experimental design. Much of the knowledge about Lake Erie is confined to a small portion in the unique western end where Ohio State University has fostered limnological research since the 1930's. Conclusions about observed changes throughout the basin are often sketchy and liberal interpretations have been made on what are undoubtedly complex ecological processes.

What has caused the concern over Lake Erie? Several manifestations of change have been observed. Briefly, the general feeling is that Lake Erie is undergoing cultural eutrophication associated with increased nutrient introduction caused by human



activity in the watershed. Beeton (1961, 1965, 1969) has summarized data gathered since 1900. He concludes that dissolved solids, including forms of phosphorus and nitrogen, have increased by more than 50 mg/l, water temperatures have increased about 1.1 C and oxygen has been depleted from bottom waters over extensive areas. Also, there have been major changes in species composition of fish and bottom fauna and increased abundances of phytoplankton and zooplankton. Carr (1962) has indicated that the depletion of oxygen is becoming more extensive. The increased depletion of oxygen is thought to result from increased decompositional demand created by production of organic carbon, this production presumably stimulated by nutrient introduction; particularly phosphorus.

Investigations of federal agencies led to publication of the Lake Erie Report (FWPCA, 1968) and the general conclusion, *"pollution in Lake Erie and many of its tributaries causes significant damage to recreation, commercial fishing, sport fishing, navigation, water supply and aesthetic values. . . . Eutrophication or over-fertilization of Lake Erie is of major concern. Problems are occurring along the lake shoreline at some water inlets and throughout the lake from algal growths stimulated by nutrients. Reduction of one or more of such nutrients will be beneficial in controlling algal growths and eutrophication."*

The most specific concerns deal with deleterious changes in resources drawn from the lake. There have been alterations in fisheries from one that produced economically desirable species to one now dominated by less desirable species. Beeton, (1961, 1969) has projected that changes in the fisheries are related to eutrophication. Although the total commercial catch has remained constant, the value of the catch has decreased. The sports fishery may have changed similarly. The contamination of fish by mercury has also been a severe aggravation.

Aquatic macrophytes and algae are considered nuisances in several ways. They accumulate on beaches and rot. This occurs primarily at the eastern end of Lake Erie. Attached algae foul docks, boat props, buoys, etc. thereby increasing maintenance costs. Phytoplankton have imparted noxious tastes to water, especially those species associated with nutrient enriched lakes,

and filamentous species have been known to clog water treatment filters. Together these conditions raise water treatment costs. Turbidity in the lake caused by suspended solids also contributes to water treatment costs and generally depreciates aesthetic values more than any other factor.

Local concentrations of solid and liquid wastes discharged along the shore are frequently associated with shoreline industry. Industry, in itself, standing on the horizon perhaps represents the greatest depreciation of aesthetic values. Health is directly threatened by untreated human wastes near municipalities as indicated by heavy concentrations of coliform bacteria. Swimming and other water use is thwarted in those shoreline localities.

In short, Lake Erie exhibits many undesirable traits that include beaches contaminated by disease organisms, aesthetically displeasing solid and liquid wastes, high turbidity, nuisance growths of aquatic plants and depreciated fishery values. If these are to be corrected, the causes should be identified. Evidence presented in this report will show that the present condition of Lake Erie is the result of a whole complex of change and that restriction of municipal and industrial waste input alone is not likely to satisfy all of the present complaints about this degraded water resource.

#### Fish Populations in Lake Erie

Change in Lake Erie fish populations is potentially the most disturbing, if not the greatest, alteration in the socio-economic status of the lake. It is amazing that so much misinformation about Lake Erie fishes has been passed to the public. Commoner (1966) has claimed that fish are nearly extinct, when in fact, Lake Erie yields as many fish to commercial fishermen as all of the other Great Lakes combined! The yield in weight has fluctuated without trend (IJC, 1969) since the last century and averaged an impressive 5 to 10 lbs per acre or about 8 times the yield from the other lakes combined. Although there has been no change in total commercial yield during this century, the species

composition has changed from one dominated by whitefishes and perchpike (Stizostedion spp.) to smelt and yellow perch. The change was undoubtedly created by man. Any number of specific causes or combination of causes have been suggested, including over fishing, accidental and intentional introduction of exotic species, habitat changes in spawning tributaries and, increased loading of the lake with sediments, biostimulatory nutrient salts, and toxins. Identification of the individual impacts and source of effect is not simple, but the available data suggest that the greatest impact on Lake Erie fishes resulted from a combination of over fishing, man-caused turbidity, sedimentation, and modification of spawning tributaries.

The early history of Lake Erie, as summarized by Trautman (1957), shows that major changes occurred in the fisheries before municipal and industrial wastes became an issue. The commercial fishery was important by 1830 but restricted mostly to shores and spawning runs up the tributaries. Dams were constructed in the tributaries for milling purposes and weirs were constructed to take fish. Spawning runs of pike, muskellunge and sturgeon were impeded long before any effects were observed in the lake and decreased abundances of these species were noted before 1850.

Clearing the watersheds for agricultural purposes progressed and tributaries became continuously turbid long before the turn of the century. Aquatic macrophytes and clear waters that favored an abundance of sight feeding fish passed from existence in the river mouths at about the same time in the late 1800's and early 1900's. Also, about 1850, the gill net and pound net were introduced to open the fishery in the lake proper. Evident decreases in fish yields caused a demand for an investigation about 1875. Artificial propagation and introduction of exotic species were offered as the solution. Among those early introductions, carp and goldfish were the only ones to attain prominence, particularly in the western basin. Trautman (1957) indicates that about 85 species of fish have been reported in the Ohio waters of Lake Erie. Of those species 43 have notably declined in abundance

and 8 have been introduced (and in several cases flourished). Five of the species that have declined greatly since 1910 have been important in the commercial catch. They include Northern pike, cisco, sauger, whitefish, and walleye (blue and yellow).

Before 1910 and before the time of much demonstrable change in nutrient input to the lake (Beeton; 1965, 1969) several other food species declined greatly enough to elicit protective laws. These species included sturgeon, muskellunge, smallmouth bass, largemouth bass, and bluegill. Trautman (1957) blamed increased turbidity for the decline of the remaining species. Many of these are "sight-feeders" or they require aquatic macrophytes for survival. In fact, Trautman (1957) thought that massive modifications in all watersheds of Ohio caused widespread depressions of many fish populations because of increased sediment loads.

No one has satisfactorily demonstrated any important contribution of industrial and metropolitan wastes to the recent decline in Lake Erie fish populations, although negative inferences have been drawn. Authors of the International Joint Commission report (1969), and others have inferred that overfishing was not the only factor that caused a decline in commercially desirable species. They cite Beeton (1965) who says that reproductive failure has also contributed to the decline but without documentation of any kind (although blue-pike reproduction ceased after 1953 according to Regier, et al.). Regier, et al. (1969) have summarized the change in the commercial fishery of Lake Erie, particularly that of walleye in the western basin. They have advanced explanation to explain changes in walleye, (yellow and blue), whitefish, and cisco abundance, that incorporates fishing pressure, introduction of smelt, and eutrophication. Their best data indicates overfishing as the main cause for the changes in the commercial fishery. In fact, that explanation alone is sufficient to explain the changes, but based on the reports of Carr (1962) and others, they have referred to the possible effects of rapidly extending oxygen depletion in the central basin. As will be discussed later in this report, the purported changes in oxygen concentration are not



demonstrable (although they may have occurred). But they suggest the possibility that progressive deoxygenation was the primary factor responsible for excluding most whitefishes and cisco from summer habitat (it is known that they moved out of the central basin in summer during the 1800's when they were most abundant in the catch) and may have been responsible for excluding blue-pike (walleye) from spawning grounds, crowding them and making them more susceptible to fishing pressure with gill nets.

Regier, et al. (1969) believe that the walleye have been progressively displaced from the hypolimnetic region of the central basin. After leaving, this left a refuge for young perch and smelt which use the bottom of the epilimnion (over the oxygen depleted regions) and has enabled greater abundances of these species to grow older and become effective predators on walleye fry. Great abundances of adult smelt were first observed in 1951. However, smelt appeared in Lake Erie for the first time in 1932 (Trautman, 1957) and it is possible that 20 years were required for the population to build to this point, in view of the fact that spawning sites available in the Erie watershed were restrictive. Alewives similarly were first discovered in 1931 but were not considered numerous until the 1950's (Trautman, 1957). Perch have probably always been relatively abundant, although it is difficult, from commercial fishing records, to demonstrate changes in abundance because of changes in fishing intensity for that species.

The purported effects of low oxygen concentrations on commercial fishes apparently have not had similar impacts on non-commercial or "deep water" species such as the smelt. Bottom species like the darters and sculpins were still reported to inhabit the cool central basin in the 1950's (Trautman, 1957). Burbot also persist, although incomplete records for this species are available. Lake trout also remain in the eastern end of the lake. It should be noted that most of the declining non-commercial fish are shallow water species associated with aquatic vegetation.

Oxygen concentrations have been implicated with fish kills, but as the IJC (1969) indicates, most fish kills included those species that are considered relatively tolerant of eutrophied situations.

Kills of commercial species are almost invariably associated with fishing activities. The present attitude about "pollution" effects on fish is summarized by the IJC (1969).

*"In summary it is probable that pollution and eutrophication have acted to limit reproduction by altering and reducing spawning areas and otherwise restricting the habitat of various fishes, particularly the cold water varieties. These changes are at least partly responsible for declines in some valuable commercial species including the whitefish, lake herring, blue pike, sauger and walleye. The production of certain fish food organisms, notably mayflies and their larvae, has been reduced. To the extent that pollution has affected blue pike and walleye, it has also upset the balance of predator and prey species. This has resulted in excessive dominance by yellow perch, smelt and alewives in the ecosystem."*

From what can be discerned from the data at hand, the changes that have occurred are much less dependant on changes in oxygen depletion than in a poorly regulated intensive fishery and the introduction of exotic species. Apparently sedimentation caused by widespread agricultural practice has also had extensive impact on Lake Erie fishes. Intensive pollution and dams in tributaries have also been influential in blocking spawning runs.

#### Plankton and Macrophytes in Lake Erie

Historical records from the 19th century suggest that emergent aquatic plants along the shores of southwestern Lake Erie were much more important than they have been during this century (Trautman, 1957). Aquatic vegetation began to dissappear from the western basin by 1900. The earliest settlers described the Maumee as a "flooded meadow with long grass," meaning aquatic vegetation with great abundances of wild rice. Apparently remarkable numbers of waterfowl were associated with these marshes with many species still persisting to this time. Details of waterfowl will be discussed later. Trautman (1957) associates the disappearance of macrophytes with the increased turbidity and sedimentation from

erosion in the tributary streams. There is probably no better explanation. Early taxonomic studies of plankton in the lake were conducted (Davis, 1964) but very few quantitative studies have been applied. Of those that could be called quantitative, variations in sampling techniques make comparisons difficult.

Phytoplankton abundances are relatively high. Davis (1964) conducted the only long term study to define changes in phytoplankton abundances and determined, fairly conclusively, that mean annual phytoplankton abundances have at least tripled between 1927 and 1964 in one sampling point near Cleveland in the central basin. His data also indicate that the main reason for the mean annual increase is that seasonal maxima have enlarged and minima have receded in the importance of annual dynamics. These changes indicate that seasonal limitations have in some way been alleviated. This data is probably representative of conditions throughout that part of the central basin according to data gathered by Powers, Jones, Munding and Ayers, (1960).

Seasonal limitations, in winter and summer, have traditionally been associated with large lake systems, the common explanation being that limiting nutrients, often phosphorus, are removed by phytoplankton during spring and or fall turnovers and then deposited in the sediments as dead algae. However, Chandler (1944) has indicated a periodicity in the western basin of Lake Erie that cannot be explained in this manner because the western basin forms no hypolimnion. Marcus (1972) indicates a different pattern for the period May through November in which numerical abundances were relatively constant except for a high in May and lows in late fall. However, in units of planktonic carbon, the peak occurred during a blue-green bloom in August. The reason for this is that the mean size of many of the summer plankton (blue-green) is larger than the spring and fall plankton (diatoms). Earlier work was not expressed in terms of biomass. Here, blooms would be much less obvious or completely different if the data were expressed in biomass units. Therefore, the biomass of flora in the central basin may be relatively constant throughout the summer and not necessarily limited

by the availability of nutrients. Some indications of species compositional changes have occurred in the central basin and western basin but these are far from obvious alterations (Davis, 1964; Verduin, 1964).

Primary productivity has been measured by investigators using several different techniques since 1948. Verduin (1964) has claimed that productivity estimates, based on changes in gas concentration (oxygen, carbon dioxide), in the open lake system doubled between the lake 1950 and 1962. Cody (1972), using the same method, found the same values in 1971 as were reported for 1962. Mean productivities based on  $C^{14}$  estimates range from lows of 8.1 to 31.0  $mgC/m^2/day$  in 1963 (Saunders, 1964) to 87  $mgC/m^2/day$  in 1968 (Parkos, et al., 1969) to 833  $mgC/m^2/day$  in 1972 (Cody, 1972). These values would suggest that productivity has increased by two orders of magnitude for the past decade rather than leveled off. Chlorophyll studies on the other hand, suggest that productivities decreased by 50% from 1948 (Tucker, 1949) to 1961 (Verduin, 1964).

In brief, the productivity estimates made for Lake Erie are not reliable and have not demonstrated anything conclusive about changes in productivity. Part of this inconsistency is due to technique, but even within one technique, as much as an order of magnitude difference can occur between stations and two orders of magnitude between two successive days. Spurts of productivity in the form of blue-green blooms have supposedly increased in frequency in the western basin (Casper, 1965), but it should also be remembered that sampling frequency has also increased.

Like phytoplankton, changes in zooplankton abundances are not easily assessed from the data available because of few quantitative studies. Wright, et al. (1955), at a station near the mouth of the Raisin River, reported mean concentration of total zooplankton for May to September 1930 to be about 20% of that reported by Nalepa (1972) in the same general vicinity. The sampling techniques (Juday trap and VanDorn water bottle) were similar enough to consider sampling differences negligible.



Apparently the concentrations have increased greatly over forty years in the western end of the lake. In the island region of the western basin, the results of Chandler (1940) and Verduin (1949) may indicate that the cladoceran and copepod population almost doubled in a decade. It is difficult to place any estimates of variability on these data because so few studies have been conducted. However, where comparisons can be made, it appears that zooplankton and phytoplankton abundances have increased rapidly over the last forty years. Changes in the taxonomic composition of the zooplankton are not discernable.

In summary, the data is patchy and sometimes questionable. But it appears that in the western basin, there has been a shift from macrophytic production to microphytic production mostly before 1900 and that standing crops of both phytoplankton and zooplankton have increased similarly. Presumably these increases have followed concurrent changes in productivity. Increased standing crops could also occur if secondary consumers (fish) had markedly decreased in predatory effectiveness.

#### Benthic Fauna in Lake Erie

Changes in zoobenthos are among the most definite to be reported from western Lake Erie. Although members of zoobenthos are of no direct socio-economic concern, except for nuisance emergents, they may have had a critical influence on fish populations through the food web. Undisputed changes in benthic populations have occurred since 1930. However, the cause of those changes is probably more complicated than just a simple response to periodic deoxygenation during temporary stratification. Therefore, alleviation of the cause of deoxygenation may not allow full recovery of the zoobenthos.

Britt (1955a, b) was the first to call attention to obvious changes in the zoobenthos. He associated a 90% reduction in the abundance of burrowing mayflies, with the depletion of oxygen from bottom waters during a period of warm, calm weather in 1953. During the following year mayflies had almost completely recovered

to their former abundance. Wright, et al. (1955) reported similar abundances in the island region in 1929 and 1930. At that time, however, the annual variation was nearly 100% but there was no apparent deoxygenation involved (although sampling intensity will control the probability of finding depletion).

Similar changes in annual abundance are also implied by Chandler (1963). Since 1954, however, the burrowing mayfly has almost entirely disappeared from Lake Erie. At the same time, severe oxygen depletions have been more frequently observed and it is logical to assume, as Beeton (1969) has, that hypolimnetic oxygen depletion has caused the zoobenthic changes. There has never been any demonstrable effect of zoobenthic change on fish populations in Lake Erie. In fact, only one definite socio-economic change has been realized - nuisance emergences of mayflies no longer occur. Some would like to keep it that way.

If, as Carr (1962) and Beeton (1969) have suggested, oxygen depletion is associated with oxygen demand in the sediments, then the distribution of organic material and mayflies should reflect the effects of this oxygen demand. Wood's (1963) discussion of zoobenthic and organic distribution in sediments of the western basin should have indicated that those sediments with the greatest organic content would have contained the least mayflies. On the contrary, from Woods (1963) data, mayflies were most abundant in the presence of organics except near the mouths of major tributaries. Abundances of mayflies were universally low near the mouths of the Detroit and Maumee Rivers, even though the organic content in those sediments ranged from less than 1% to more than 4% just as in the lake proper.

Carr and Hiltunen (1965), Wright, et al. (1955) as well as Wood (1963) all indicate the strong distributional orientation of zoobenthos to the major tributaries. Increased abundances of oligochaetes, in particular, suggests increased organic enrichment about river mouths from 1930 to 1960 (Carr and Hiltunen, 1965). The data of Carr and Hiltunen also suggest that this effect on oligochaete distribution has been immense; polluted areas, based

on oligochaete numbers, were extended by at least ten times from 1930 to 1960. Presumably most of the organic input came from municipalities which had merely doubled in size during that time.

Therefore, the change brought about in zoobenthos was probably not due strictly to a BOD release to the lake. Mayflies particularly have been very susceptible to effluents from the tributaries. This was true in 1930 (Wright, *et al.*, 1955) at the mouth of the Detroit River, even though oxygen demands were negligible (unlike the mouths of the Raisin and Maumee Rivers which often approach anoxia). It might be argued that progressive changes in zoobenthos have been due to an ever increasing fallout of phytoplankton as a result of eutrophication. But even the inputs of biostimulants does not satisfactorily answer apparent rate of the change in the distributional patterns because, as Hentley and Potos (1971) indicate, there does not appear to be that much difference in the biomass of phytoplankton throughout the basin. It should also be remembered that short term periods of intensive oxygen depletion have been described (Carr, Applegate, Keller, 1965; Britt, 1955a; Britt, *et al.*, 1968) only outside the region indicated to be moderately to heavily polluted. This does not mean that oxygen depletion may not have occurred in those areas. It has simply never been reported. Data from Wood (1963) for benthic distribution in 1953 compared to 1930 and 1961 (Carr and Hiltunen, 1965) suggests that the rate of change was greatest between 1953 and 1961, but only on the basis of mayfly distributions. If simple organic enrichment is not solely responsible for changes observed, then what is? Perhaps no one factor. But the output of industrial organic wastes and other materials have soared since World War II. Mercury has been released into Great Lakes waters for at least a half of a century. Virtually nothing is known about the combined effects of oils, pesticides, industrial additives and other wastes within the sediments. Apparently, all three of the major areas used by mayflies in the Great Lakes (western Lake Erie, Saginaw Bay and Green Bay) have exhibited decreases in abundance during the 1940's and 1950's (Beeton, 1969). It is

significant to note that these are all shallow basins which have a history of receiving heavy industrial and municipal wastes.

#### Oxygen Depletions in Lake Erie

Beeton (1969) summarized the sentiments of several investigators who feel that, among a number of changes observed in the Great Lakes, the one with the greatest impact has been the change in the sediment caused by "*tremendous amounts of allochthonous materials entering the lake.*" These changes are indicated by extensive depletion of dissolved oxygen and major changes in the species composition of zoobenthos. Beeton (1969) reiterates the conclusions of Carr (1962), Carr, Applegate and Keller (1965) and Britt, Skotch and Smith (1968) that: (1) sediments are responsible for observed oxygen depletion and (2) the oxygen demand has been increasing. The primary purpose of this section on oxygen in Lake Erie is to demonstrate that there is no good direct evidence that oxygen depletion is increasing and that sediments are not primarily responsible for the demand that does exist. These are important considerations because the belief in rapidly increasing oxygen depletion has been an important force behind "cleaning up Lake Erie."

Change in Intensity of Oxygen Depletion: Little was known about oxygen concentrations in the lake before 1928-30 when Wright (1955) and Fish (1960) conducted their limnological surveys of the lake. Beeton (1966; 69) Britt, et al., 1968, Carr, 1962 and Carr, et al. (1965) all have inferred increasing oxygen demands from data that does not warrant that interpretation. Britt (1955a) was the first to report strongly depressed oxygen in the western basin during an atypical stratification. At that time in 1953 Britt measured an average concentration of about 1 mg/liter oxygen near the bottom for three days. There was no indication of a trend in oxygen change during those three days. The observations were made at the end of a month of calm warm weather. Stratification could have existed throughout that calm spell or have been set up anytime during the period. Earlier observations in the

area by Wright, et al. (1955), Chandler (1940, 1941, 1942) indicated no exceptional depletions of oxygen (80-90% at bottom) and also no indication of stratification [at least 3° C difference from surface to bottom as indicated by the data of Britt, et al. (1968)]. Not finding stratification was probably due to a relatively low sampling frequency and chance. However, Carr, et al. (1965) also discovered a temporary stratification and associated oxygen depletion in the western basin during 1963. Because oxygen was depleted to 1 mg/liter within 5 days of preceeding calm weather (rather than the 28 days of calm it apparently took in 1953), they concluded that oxygen demands probably have increased. Britt, et al. (1968) measured temperature and oxygen over the summer and found a similar rate of decrease during a temporary stratification (as that observed by Carr, et al. (1965)) and concluded that this reinforced the Carr, et al. belief that oxygen demands have been increasing in the western basin. But, before Britt's (1965a) observations, oxygen concentrations could have stabilized at 1 mg/liter (as observed) after only a few days and then remained at that concentration until measured by Britt over three weeks later. In fact, observations made in the central basin by Carr (1962) in 1961 indicate that oxygen concentrations reach a relatively low level, near 1 mg/liter, within two to three days after stratification, and then varies about that low concentration until mixing resumes. There is no reason to believe this could not have occurred as far back as 1930 since no stratification was observed then to prove it one way or the other. Yet, if the bottom oxygen concentrations are compared for 1930 (Wright, et al., 1955) and 1966 (Britt, et al., 1968), with the periods of observed stratification deleted, the values compare closely. In fact, the oxygen demand of western Lake Erie is high enough in summer to depress oxygen concentrations by 10 to 20% when the lake is homothermous. It seems that mean demand at the bottom during mixing has not changed much since 1930.

There is also very inconclusive evidence of change in oxygen demand in the central basin as suggested by Carr (1962). He also



recognized that annual variation in the extent of oxygen depletion was two to three fold. The western edge of the central basin, which had the most intensive depletion of oxygen in 1961 (down to about 1 mg/liter), also had the same concentrations during stratification in 1930 (Wright, 1955). Wright also indicated that this depletion could have occurred in less than a week (from 4.8 mg/liter on 5 August to 0.78 mg/liter on 9 August) from data he had obtained 4 days earlier at a station 7 miles away. These observations indicate that the potential for rapid and intensive depletion of oxygen existed in 1930 in the west end of the central basin. Fish (1960) also obtained data farther north and east in the central basin on the periphery of the area which Carr in 1961 recognized as intensively depleted of oxygen (1 mg/liter or less). Fish (1960) showed that depletion to 42% of saturation could occur within an unknown period of stratification. It should also be emphasized that stratification is not stable during the summer in the central basin (Carr, 1962; Fish, 1960; Wright, *et al.*, 1955). Therefore, measurements have left much to chance and do not reveal anything about change in oxygen demand over the last half century. Only one point is certain, oxygen depletion to very low concentrations occurred as early as 1930 in the central basin.

In the eastern basin, oxygen has never been observed to fall below 5.5 mg/liter in the hypolimnion. Slight changes in the depletion rate of oxygen in that basin are possible (Carr, 1962). The trend is present, but wide annual variability again casts doubt on the reliability of any trend obtained from a few data points taken over a thirty year period.

Source of Oxygen Depletion: Oxygen is definitely being depleted from the hypolimnion of the western and central basin of Lake Erie when stratification occurs. This demand may be from (1) material suspended or dissolved in the water or (2) the sediments. The material may be derived from (1) autochthonous (intrinsic) or (2) allochthonous (extrinsic) sources and it may exert (1) chemical or (2) biological demand. Beeton (1969) thinks that demands are primarily due to the introduction of allochthonous

materials in the form of municipal and industrial wastes which have accumulated in the sediments, and that sediments rather than suspended matter exert the major demand for oxygen. The general premise to be presented here is that these hypotheses are not demonstrable and are probably less likely than those which suggest that oxygen demands originate mostly in the water rather than from sediments.

Measurement of the carbon content in Lake Erie sediments indicate that carbon concentrations are not particularly high compared to the other Great Lakes (Skotch and Britt, 1969; Callender, 1969; Kemp, 1969). Also carbon/nitrogen ratios are very similar among Great Lake sediments including those from the oligotrophic Lake Superior to the eutrophic Greenbay of Lake Michigan and western Lake Erie. There is no sign that organics in sediment from the main western basin have the higher C/N ratio that is sometimes associated with wastes. The carbon and nitrogen in all the sediments appears to be primarily derived from phytoplankton input. Comparisons of the amount of carbon introduced to Lake Erie as BOD amounts to 70-100,000 metric tons/year (IJC, 1969). On the basis of a mean concentration of 10 mg/liter organic carbon found by Cole (1972) the mean annual carbon load in the western basin is 210,000 tons. The theoretical flow-through time in the western basin is about 2 months. Therefore, based on that flow rate and removal rate of carbon alone and without consideration of decay processes, the introduction of allochthonous carbon can amount to only 5 to 10% of the total organic carbon fixed photosynthetically. These are undoubtedly rough estimations and probably underestimate the role of photosynthesis, because carbon turnover rates within the 2-month, mean, flow-through time are not included in the calculation.

There are also indications that the high oxygen demand reported by Beeton (1969) is about an order of magnitude higher than that found by Skotch and Britt (1969). The latter authors found that this oxygen demand was not well correlated with the organic carbon content of the sediments. It may have been better correlated with iron content in the sediments. If so, this would agree with the

theoretical discussion by Potos (1970) who suggests that both chemical and biological oxygen demand operate under stratification following resuspension of the bottom sediments. The chemical demand could come from reduced forms of iron and sulfur. But, oxygen depletions in the western basin are not likely to be due primarily to chemical demands. When stratification occurs, sediments would have to be resuspended by improbable internal seiches, during exceedingly calm periods. The observed rates of oxygen depletion after stratification, about an average of 1 mg/l/day over five days, can be accommodated by the respiratory demand measured in Lake Erie water by Cole (1972) and Cody (1972). Suspended and dissolved organics can remove oxygen both as they settle and as they lay in a relatively thin active surface-layer on the bottom. Undoubtedly resuspension of anaerobic sediments in aerated water in laboratory experiments will cause a rapid depletion of oxygen because of chemical demand. However, that is an unlikely source of oxygen depletion in western Lake Erie.

The vertical distribution of carbon in the sediments of Lake Erie and other Great Lakes with the exception of western Lake Erie and Lake Superior indicates an accelerated introduction of carbon and nitrogen in relation to inorganics in the sediments (Kemp, 1969, Callender, 1969). In Lake Superior no change has occurred in sediments in recent years but in the western basin of Lake Erie, the surface sediments (top 20 cm or approximately 50 years accumulation) have lesser concentrations than subsurface sediments which contain plant debris much like small lakes (Kemp, 1969). This corroborates reports of 19th century historians who described extensive shallow areas dominated by aquatic macrophytes. Increased inputs of inorganics, particularly from the Maumee watershed also dilute the organic sediments. In the rest of Lake Erie, Lake Michigan and Lake Ontario organics are between 30 and 50% higher in the surface 10-20 cm (Kemp, 1969; Callender, 1969). These differences would be even greater if corrections were made for increased inorganic sedimentation in these lakes, particularly in

Lake Erie. Although there has been no direct evidence of increased primary productivity in Lake Erie as discussed previously, the vertical distribution of organics in the sediments suggests that greater quantities of algal organics are now reaching the sediments compared to a century or more ago. This could reasonably result from increased productivity, decreased consumer efficiency (primarily zooplankters) or a combination of the two. Just how much consumer efficiency has changed is impossible to assess because adequate measurements of primary production and consumption have not been made. As already indicated it appears that both zooplankton and phytoplankton have increased in western and central Lake Erie so productivity has probably increased there somewhat. Of course, within the surface layer of sediments, it is impossible to define gradients in carbon concentrations because of the mixing of sediments by wave action and animals. Therefore, sedimentology does not reveal the rate at which organic input into the sediments has been changing since the turn of the century.

#### Dissolved and Suspended Solids in Lake Erie

Dissolved and suspended solids have undoubtedly increased in Lake Erie (Beeton, 1965; Trautman, 1957). Meagre records of suspended solids exist but transparency, which is determined primarily by concentrations of suspended solids in Lake Erie (Verduin, 1964), apparently were markedly altered in the western basin toward the end of the nineteenth century (Trautman, 1957) based on the reports of several independent observers. Increased turbidity coincide with draining and farming the swampy, Maumee watershed. The soils in that region, comprised mostly of clay, are particularly susceptible to erosion.

Transparency (and probably suspended solids) has not changed demonstrably since the first secchi-disc observations were made in the early 1900's (Beeton, 1961). Transparency has remained at 1 to 2 m during that time. Agricultural activity has also remained stable while the human population quadrupled. Presently, more than half of the sediments introduced into the western basin are



carried by less than 5% of the inlet waters via south-shore streams (IJC, 1969). Transparency varies directly with distance from the south shore (Pinsak, 1967). Variations in concentration of suspended solids moving down the Raisin River follows discharge and indicates a diffused origin throughout the watershed. The wind has been blamed for much of the turbidity because the basin is so shallow (Beeton, 1961), but this appears to be a secondary cause that has been dependent on the influx of fine sediments following widespread disturbance in the watershed by agriculture. According to data reported by Cole (1972) and Marcus (1972) plankton are minor contributors to turbidity except during summer blooms.

Increases in total dissolved solids are very well documented by Beeton (1966, 1969). Also, comparison of data from Lake Huron and the other Great Lakes indicate a more rapid accumulation of total dissolved solids than is explainable simply by natural watershed contribution. Total dissolved solids have increased by about 60 mg/liter, or 43% since the last century (Beeton, 1969). About half of that increase is explained by increases in chloride and sulfate. The ratio of increase for  $\text{Cl} : \text{SO}_4 : \text{Ca} : \text{Na} + \text{K}$  is equal to 3.2: 2.6: 1.6: 1.0. Ratios found in sewage are 0.5: 0.3: 0.5: 1.0. Total dissolved solids in sewage average about 2.5 to 3 times the concentration of combined sodium and potassium while in Lake Erie total dissolved solids have increased by 12 times the amount of sodium and potassium. It is likely that at least 75% of the increase in dissolved solids has come from sources other than municipal wastes.

In the late 1800's the Lake Erie watershed contributed measurably, about 33%, to the total dissolved solids found in Lake Erie according to data presented by Beeton (1969). Because the Lake Erie watershed only contributes about 12% of the flow through the basin, mean concentrations of dissolved solids in small tributaries had to be relatively high (about 435 mg/liter). In fact, this value is similar to values from highly disturbed watersheds reported at the present time (unpublished data M.D.N.R.) but greater than those reported by Dole (1926) for streams in the area



around 1906-07. This suggests that Beeton's (1969) early data may not be an accurate estimate of concentrations in Lake Erie and Lake Huron. However, there is little doubt that increases have occurred.

The environmental impact of small changes (less than 100%) in dissolved solids is unpredictable without knowledge of the kinds of solids that have changed concentrations. Certain relatively rare nutrient elements may be the key to an array of biological and physical changes in lakes that are generally embodied in the term eutrophication. Although a number of elements and other physical conditions may limit algal growth and various associated environmental alterations, phosphorus has been most frequently implicated as the limiting nutrient in fresh waters. In estuarine and marine water, nitrogen is frequently the limiting element. Carbon has been identified as limiting where alkalinities and organic inputs are very low as they are in some waters in the southern United States or in very eutrophic situations where growth can outstrip the capacity of the system to supply  $\text{CO}_2$ . Sometimes light may be limiting to algal growth and wind has also been implicated. Of course any of a number of toxic materials can limit the growth of algae in large enough quantities.

Some of the work conducted by Verduin (1960) has indicated a possible  $\text{CO}_2$  limitation. He observed that the diurnal rate of  $\text{CO}_2$  change (reflecting photosynthesis) decreased before optimum light levels were reached. King (1970) has indicated that equilibrium levels of  $\text{CO}_2$  are likely to become limiting when pH exceeds 9 in a lake such as Lake Erie with total alkalinity near 100 mg/liter. Verduin (1964) has reported pH up to this value in the island region of western Lake Erie. It is therefore possible that western Lake Erie is nearing carbon dioxide limitation during summer months at the surface.

Undoubtedly carbon inputs have increased in the western basin over the last century. However the carbon input to western Lake Erie, the equivalent of 200,000 tons/year  $\text{BOD}_5$  is presently discharged into the lake (IJC, 1969). This represents a carbon equivalent

of 75,000 metric tons, and is less than 0.1% of the bicarbonate carbon that enters the lake via tributaries. Marcus (1972) has reported that mean concentrations of phytoplankton carbon range between 0.5 to 1 mg/liter or 2.5 to 5% of the bicarbonate carbon present. This indicates that the 25 to 50 times more carbon is fixed in algae than is made available through waste loading. But, waste carbon can be relatively important near major discharge points, such as the Raisin River (Marcus, 1972). Although the total increase in carbon from waste contribution has been minor, the form of carbon may be a meaningful criteria for availability.

Increases in phosphorus concentration over the last three decades have been reported by Verduin (1964), IJC (1969) Hentley and Potos (1971) and Harlow (1968) although inadequacies in early analytical techniques and insufficient evaluation of spatial and temporal variation devalue these data. If the data of Beeton and Chandler (1963) can be judged adequate, Lake Superior phosphorus concentrations (total P) have been on the order of 5  $\mu\text{g/liter}$ . Lake Michigan and Huron are about twice that amount and probably always have been because of differences in the geology of their watersheds. Data on phosphorus from relatively undisturbed streams in lower Michigan (Haines, Personal communication) indicate that concentrations of 5 to 10  $\mu\text{g}$  of total phosphorus were probably typical in Michigan streams before human influence became widespread. Streams in the Lake Erie watershed run greater than an order of magnitude higher than that value at the present time and partially contribute to the large increases in total phosphorus as water passes from Lake Huron to western Lake Erie. Assuming that concentrations in western Lake Erie were originally similar to Huron, total phosphorus has increased about 6 fold during the last century. More important, if the data of Chandler and Weeks (1945) can be accepted as representative, most of this increase has occurred during the past three decades. In fact from the summary given in IJC (1969) phosphorus concentrations quadrupled during the last three decades. During the same three decades, total nitrogen has only tripled but ammonia has apparently increased

by an order of magnitude in the western basin since 1930 (IJC, 1969) compared to only a doubling in the central basin.

The rapid increase in phosphorus input to the lake reflects increasing use of fertilizers, use of detergents and increase in human wastes. In fertilizers phosphorus application in the United States doubled between 1950 and 1965. During about that same time the population itself increased by less than 50% so the contribution from human waste was relatively small. But the per capita contribution of phosphorus tripled during that time. Undoubtedly, the rate of phosphorus released to Lake Erie sharply changed following World War II because of accelerated uses in detergents and fertilizers. Total nitrogen has not increased at the same rate as phosphorus as expected because only two major outside sources exist; municipal wastes and fertilizers. As with phosphorus, changes in the human population must have caused only minor changes. Nitrogen application rates however quadrupled from 1950 to 1965 and increases in the lake are probably derived primarily from this source. According to IJC (1969) less than 45% of the phosphorus and 20% of the nitrogen is from the Detroit-southeast Michigan area. About 30% of the phosphorus enters Lake Erie via runoff and from Lake Huron. This estimate however must be crude at best and will be discussed later in this report.

In summary, although estimates are imprecise, it is probable that changes in the Lake Erie concentrations of all nutrient elements have been accelerated by human activity in the watershed. These changes are associated with increased population density, the advent and use of detergents and accelerated application of fertilizers.

#### Waterfowl in the Western Lake Erie-Lake St. Clair Basin

As was mentioned earlier, historical records from the 19th century indicate that tremendous numbers of waterfowl were associated with the emergent aquatic plants along the shores of Lakes Erie and St. Clair. Many of these still persist in spite of human insults to their habitat. The species of waterfowl

found in the western Lake Erie-Lake St. Clair regions of Michigan belong to one of four general groups: geese and swans, dabbling ducks, diving ducks, and mergansers. Kortwright (1953) reports that 44 species of swans, geese and ducks belonging to the Family Anatidae are found in North America. At least 25 species can be found in the Lake Erie-St. Clair region during part of each year (Table 1). The most common members of each group are the Canada Goose and Whistling Swan of the Geese and Swans; Mallard, Black Duck, Pintail, Green-winged Teal, Blue-winged Teal, and American Widgeon of the Dabbling Ducks; Redhead Canvasback, Greater Scaup, Lesser Scaup, Common Goldeneye, Bufflehead, and Ruddy Duck of the Diving Ducks; and Common Merganser of the Mergansers.

There are four major periods of use by waterfowl during the year. Fall migration takes place from mid-September to late December while the wintering period extends from late December to late February. Spring migration extends from late February to early May with the summer or breeding period extending from early May to mid-September (Hunt, 1957; Kelley, *et al.*, 1963; Reed, 1971).

Although there are always some waterfowl species present on the area throughout the year, the greatest number of individuals and species are present during the spring and fall. The most complete data concerning migratory movements and numbers of waterfowl are available for the fall period. Bellrose (1968) estimates that 17.5 million ducks migrate through the United States east of the Rocky Mountains each fall. He indicates that at least 4% of these ducks (680,000) use the Chesapeake Bay Corridor which includes Lake St. Clair, the Detroit River, and Lake Erie. This area is considered to be the principle corridor for the passage of diving ducks, approximately 10% of the divers migrating east of the Rocky Mountains, to Chesapeake Bay (Figure 5). The area is used by 15% (250,000) of the Scaup, 44% (130,000) of the Canvasback and 6% (50,000) of the Redheads migrating east of the Rocky Mountains. Bellrose (1968) estimates that at least 125,000 dabbling ducks, about 1% of the total that migrate east of the Rocky Mountains,

Table 1. The seasonal distribution of waterfowl species in the  
Lake Erie-St. Clair regions of Michigan.<sup>1</sup>

Waterfowl Group	Seasonal Distribution <sup>2</sup>		
	Summer Residents	Transients	Winter Visitors
Geese-Swans	Canada Goose	Whistling Swan Canada Goose Snow Goose	Whistling Swan Canada Goose
Dabbling Ducks	Mallard Black Duck Pintail Blue-winged Teal Wood Duck	Mallard Black Duck Gadwall  Pintail Green-winged Teal Blue-winged Teal American Widgeon Shoveler Wood Duck	Mallard Black Duck Pintail
Diving Ducks	Redhead	Redhead Ring-necked Duck Canvasback Greater Scaup Lesser Scaup Common Goldeneye Bufflehead Oldsquaw White-winged Scoter Ruddy Duck	Redhead  Canvasback Greater Scaup Lesser Scaup Common Goldeneye  Bufflehead Oldsquaw
Mergansers		Hooded Merganser Common Merganser Red-breasted Merganser	Common Merganser Red-breasted Merganser



Table 1 (con't.)

<sup>1</sup>Based on data from Hunt, 1957; Kelley, 1966; Kelley, et al., 1963; Pirnie, 1935; Reed, 1971; Reed and Prince, 1972; Wood, 1951.

<sup>2</sup>Season distribution based on definitions of Pettingill, 1970.  
Summer Residents: Species in an area during the summer, coming from the South in the spring to breed and returning in the fall.  
Transients: Species stopping temporarily in an area during their northward migration in the spring and during their southward migration in the fall.  
Winter Visitants: Species in an area during the winter, having come from their northern nesting grounds to pass the winter in less rigorous climate and departing north in the spring.

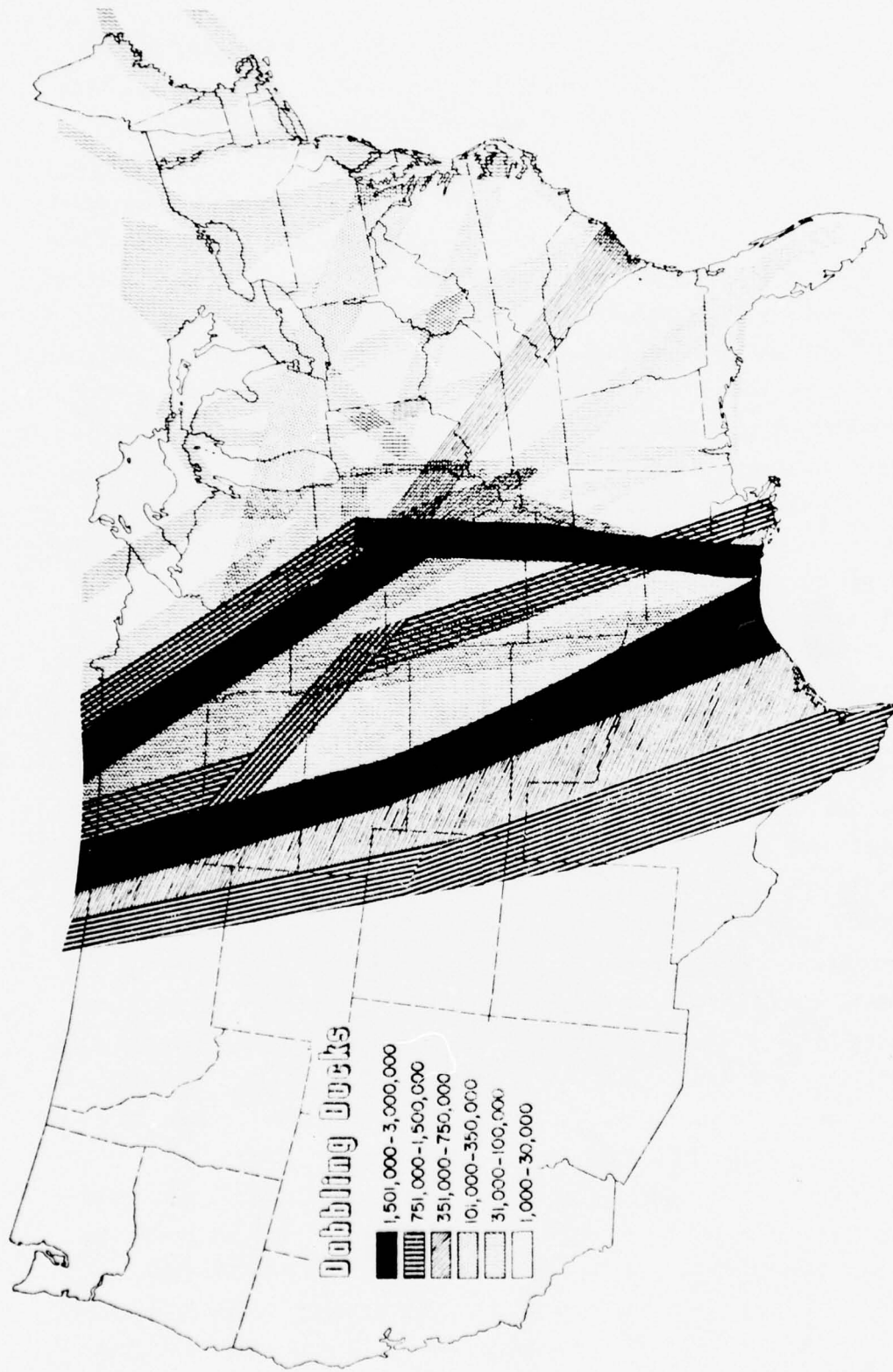


Figure 5. The fall migration patterns of dabbling ducks in the United States east of the Rocky Mountains (from Bellrose, 1968).

use the Chesapeake Bay Corridor (Figure 6). This includes 65,000 Mallards, 35,000 American Widgeons, and 25,000 Pintails. The Lake St. Clair area is used by as many as 75,000 Canada Geese in the fall (Figure 7) (Bellrose, 1968).

With an estimated 680,000 ducks using the Chesapeake Bay Corridor, the number of individuals on the area during the period of use should be less. A 19-year mean of 290,000 ducks gives some indication of the intensity of the daily use during the fall for the Michigan portions of Lake St. Clair, Detroit River, and the adjacent waters of Lake Erie (Table 2). This amounts to an estimated 20.5 million days of duck use or a period of 70 days when the population numbers 290,000 birds. Total waterfowl use, including ducks, geese, swans, and coots, amounts to about 21.8 million days. Similar numbers of ducks use the area during the spring with the 19-year daily average being 300,800 birds. The total days of bird use decreases from the fall period by about 40% to about 12.5 million days or a period of 42 days when the population numbers 300,800 birds.

The pattern of use on the Lower Detroit River differs between the spring and fall periods. The daily averages are 175,000 ducks in the fall and 250,000 ducks in the spring for Lake St. Clair while the trend is reversed on the Lower Detroit River with the daily averages being 115,000 ducks in the fall and 51,000 ducks in the spring. The larger number of ducks for a shorter period on Lake St. Clair in the spring results in a similar number of total days of bird use for both seasons of 10 million days. The total days of bird use on the Lower Detroit River decreases from 10 million days in the fall to 2.5 million in the spring. However, decline is offset by use during the winter by a mean number of 43,000 ducks resulting in 3.2 million days of bird use (Table 2).

Ducks have been present on the Lower Detroit River and western Lake Erie during the winter since the advent of open water in the early 1930's (Hunt, 1957). Mean counts of 40,500 birds for the six-year period of 1967 through 1972 are similar to the mean counts of 43,000 birds for the 19-year period of 1948 through 1966 (Table 2, 3).

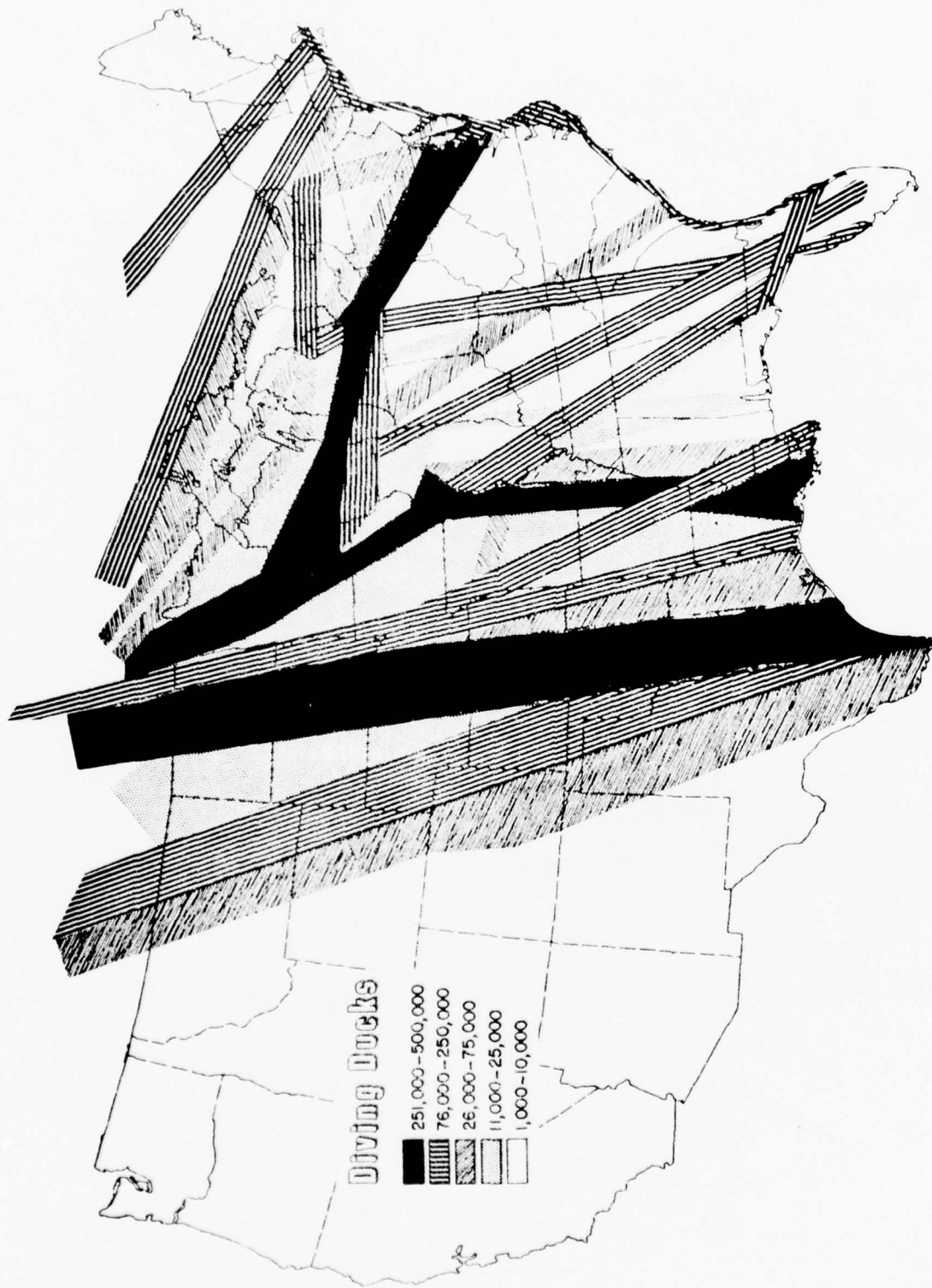


Figure 6. The fall migration patterns of diving ducks in the United States east of the Rocky Mountains (from Bellrose, 1968).

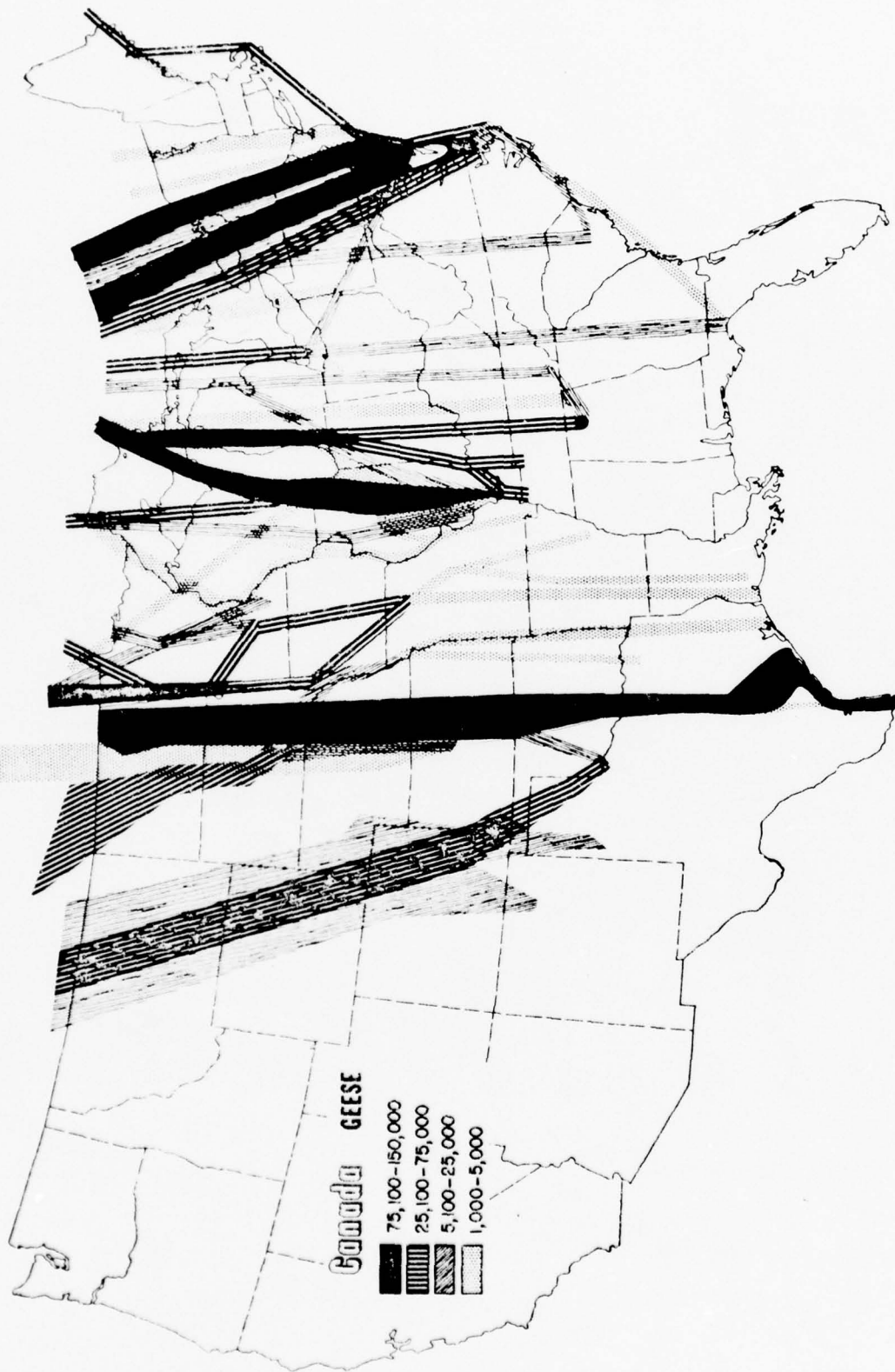


Figure 7. The fall migration patterns of Canada Geese in the United States east of the Rocky Mountains (from Bellrose, 1968).



Table 2. Average waterfowl use<sup>1</sup> of the Michigan part of Lake St. Clair, Lower Detroit River and adjacent waters of Lake Erie from 1948 through 1966 (from Mikula, 1967).

Waterfowl Group	Season			
	Fall (8/15 - 12/30)		Winter (1/1 - 3/15)	
	Average Number of Birds	Total days of Bird Use (Millions)	Average Number of Birds	Total days of Bird Use (Millions)
Ducks	290,000	20.447	43,000	3.182
Geese	8,500	0.204	-	-
Swan	4,000	0.048	-	-
Coots	25,000	1.100	-	-
			300,800	2.500
			18,800	0.375
			16,000	0.315
			14,000	0.430

<sup>1</sup>Data based on aerial census transect counts with an average of 6 during the fall period, 1 during the winter, and 2 during the spring.

Table 3. Total number of waterfowl and percentage by group or species wintering on the Lower Detroit River Lake Erie, and Lake St. Clair from 1967 to 1972.<sup>1</sup>

	Year					
	1967	1968	1969	1970	1971	1972
Total number counted	51,400	54,400	46,200	34,900	31,400	24,700
						40,500
Percentage by group or species						
Dabblers <sup>2</sup>	2	8	6	12	20	13
						10
Divers	97	91	93	85	68	79
						86
Redhead	1	1	5	1	25	2
						6
Canvasback	18	17	6	10	17	6
						12
Scaup	75	69	59	58	16	61
						57
Goldeneye	3	2	15	14	10	10
						9
Others	0	2	8	2	0	0
						2
Mergansers	1	1	1	3	12	8
						4

<sup>1</sup>Data from the annual winter waterfowl survey, Michigan, Zone 7, conducted by the Bureau of Sport Fisheries.

<sup>2</sup>Mallard and Black Duck

Although the number of birds have varied from winter-to-winter, approximately 26% with respect to the mean, the number of birds has remained relatively stable for the past 25 years. Hunt (1957) reported that the species composition for the period of 1948 through 1955 was 41% Scaup and 40% Canvasback. This has shifted in recent years to 57% Scaup and 12% Canvasback, thus documenting the continental trend of the Scaup becoming more numerous and Canvasback becoming less numerous. The most numerous species are Scaup, Canvasback, Common Goldeneye, Mallard, Black Duck, Redhead and Common Merganser.

Seven species, representing three groups, are present during the summer (Table 1). Although the lowest number of waterfowl probably occur during the summer months, population estimates are not available for this period. The open water areas are void of birds with use restricted to marsh areas bordering the lakes. The dabbling duck group is best-represented with Mallards, Black Ducks, Blue-winged Teal and Wood Ducks being the most common species.

The Lake St. Clair, Detroit River, western Lake Erie area is an important habitat component for migratory and wintering waterfowl. There are breeding birds utilizing the area, but this is of secondary importance when compared to the migratory and wintering functions.

#### Biota of Southeastern Michigan Tributary Streams

The availability of biological data varies widely for the southeastern Michigan's tributary streams. It is quite reasonable, however, to extend general information regarding these streams since the overall geographical area is small resulting in similar geological conditions.

Surveys of bottom fauna are ordinarily only undertaken by state regulatory agencies to substantiate or to determine polluted conditions. Examples of such surveys were available on the Raisin River and the Black River. Fortunately these two streams represent the southernmost (except for the Maumee) and northernmost streams under consideration (Figure 8). Thus the upstream

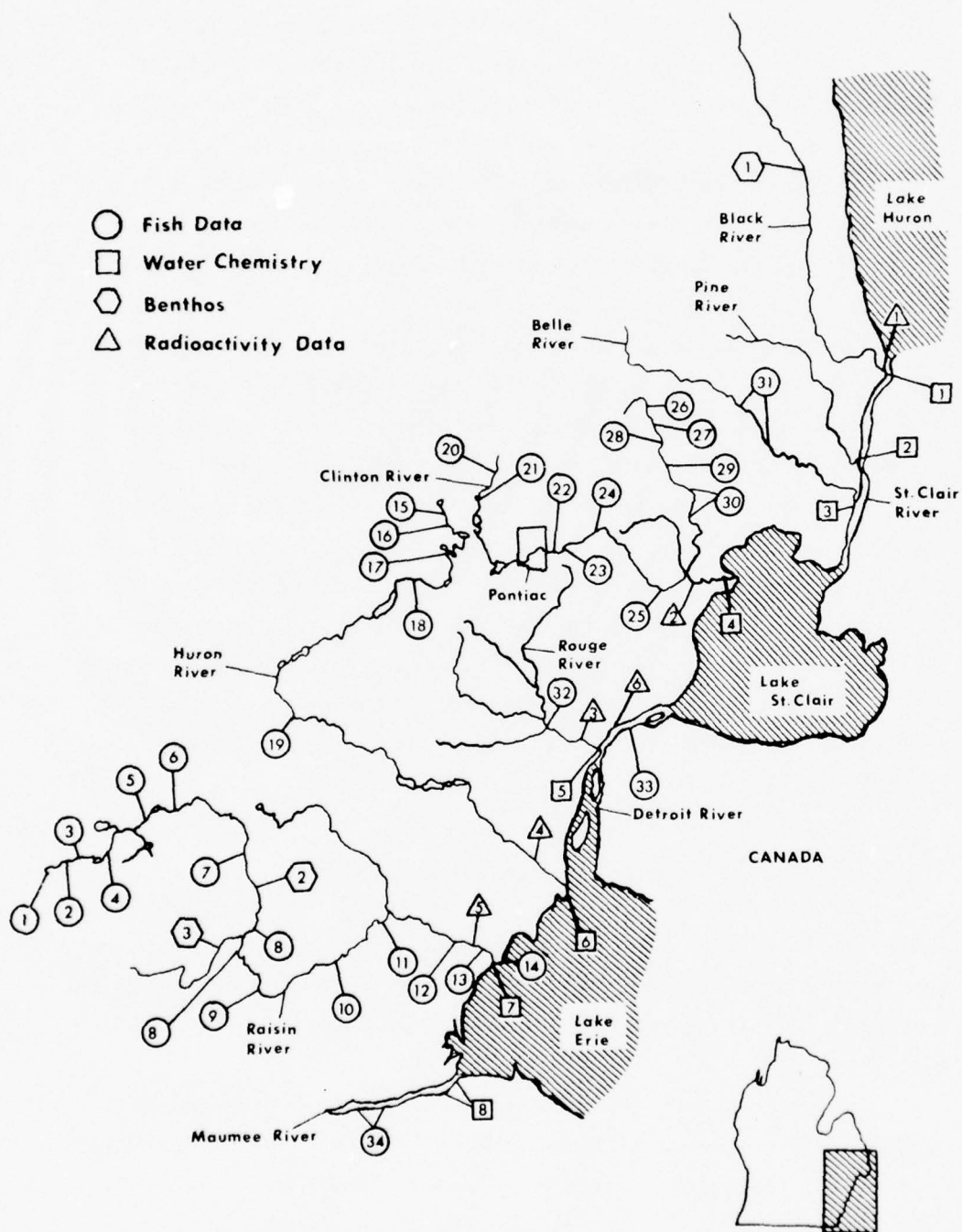


Figure 8. Sampling points on streams in southeast Michigan.

stations (above outfalls) may be considered as typical of "clean" water areas of the streams in the study area. The St. Clair River and Detroit River are, of course, exceptions because of their depth and discharge.

Tables 4 and 5 represent benthic populations in the Raisin River. In Table 4 it is apparent that water quality had decreased from 1964 to 1967 both above and below the industrial outfall being investigated. The sample above the plant in 1964 can be considered fair since the total species numbers reported is 21 which compares favorably with total numbers (29) above a different outfall in Table 5. Data from the Black River are given in Table 6 and also reveal a similar number of species (24) above the outfall. Both Tables 5 and 6 indicate the ability of streams to recover from pollution inputs; however, in each case several miles of stream are sacrificed.

These examples illustrate several points. Taxonomic lists may only be as good as the biologist examining the samples. In both Tables 5 and 6 most of the benthic organisms are keyed only to genera. Some important numbers of species may be missed in the process and likewise, important differences within a genera may exist at the species level. While these points are made it is still easy to identify gross differences that occur in the data presented.

These data reveal that it is probable that the Black River, on the basis of intolerant species present, is in better condition than the Raisin River. Also it is very apparent that the Raisin River, some miles upstream, is in far better condition than at the mouth.

No such detailed lists were available for other streams; however, the Rouge River was reported as having large crayfish populations widespread in the stream. Headwater areas had populations of mayflies and blackflies indicating cooler and better water than at the mouth. In the middle reaches of the Rouge dense populations of caddis and midges coexisted also indicating better water quality than at the mouth.



Table 4. Changes in species composition noted in 1964 and 1967 from two sampling sites above and below the outfall of an industrial outfall near Adrian, Michigan, on the Raisin River (Station 3, Figure 8).

	1.6 miles above plant		1.2 miles below plant	
	1964	1967	1964	1967
Total Number of Species	21	21	22	10
Number Tolerant Species	4	2	3	1
Number Facultative Species	13	9	14	8
Number Intolerant Species	4	1	6	1

Table 5. Benthic animals found in qualitative samples during a biological investigation of the Raisin River above and below the Tecumseh Products Company on June 18, 1968, (Station 2, Figure 8).

Organisms	2000 ft above plant	800 ft below plant	5.6 miles below plant
TOLERANT:			
<u>Tubificidae</u>	X	X	X
<u>Physa</u> sp.	X		X
<u>Procladius</u> sp.	X		
<u>Pisidium</u> sp.	X		
<u>Helobdella stagnalis</u>	X		
FACULTATIVE:			
<u>Polypedilum</u> sp.	X	X	
<u>Caenis</u> sp.	X	X	
<u>Hydropsyche</u> sp.	X	X	
<u>Cricotopus bicinctus</u>	X		
<u>Corixidae</u>	X		X
<u>Orconectes</u> sp.	X		X
<u>Hydroporus</u> sp.	X		X
<u>Gerris</u> sp.	X		X
<u>Stagnicola</u> sp.	X		X
<u>Baetidae</u>	X		
<u>Tendipes plumosus</u>	X		X
<u>Calopsectra</u> sp.	X		
<u>Microtendipes tarsalis</u>	X		
<u>Pentaneura</u> sp.	X		
<u>Antocha</u> sp.	X		
<u>Cheumatopsyche</u> sp.	X		
<u>Stenelmis</u> sp.	X		
<u>Helophorus</u> sp.	X		
<u>Planariidae</u>	X		
<u>Sialis</u> sp.	X		
<u>Ferrissia</u> sp.	X		
<u>Coenagrionidae</u>			X
<u>Trepobates</u> sp.			X
<u>Aeschnidae</u>			X
<u>Laccophilus</u> sp.			X
INTOLERANT:			
<u>Calopteryx</u> sp.			X
<u>Psephenus herricki</u>	X		
<u>Gammarus</u> sp.	X		

Table 5 (con't.)

<u>Stenonema</u> sp.	X		
TOLERANT	5	1	2
FACULTATIVE	21	3	10
INTOLERANT	3	0	1

Table 6. Benthic animals found in qualitative samples during a biological investigation of the Black River above and below the Michigan Sugar Company on November 8 and 9, 1968 (Station 1, Figure 8).

Organisms	6 miles above plant	525 yds below plant	6.6 miles below plant
TOLERANT:			
<u>Tendipes plumosus</u>		X	X
Hirudinea		X	X
Tubificidae	X		
<u>Physa</u> sp.	X		X
<u>Sphaerium</u> sp.	X		
<u>Pisidium</u> sp.			X
Naididae			X
FACULTATIVE:			
Corixidae	X	X	X
Tendipedidae	X		X
<u>Laccophilus</u> sp.	X		
Elmidae	X		X
<u>Hyalella azteca</u>	X		X
<u>Ferrissia</u> sp.	X		X
<u>Simulium</u> sp.			X
<u>Caenis</u> sp.	X		X
<u>Orconectes</u> sp.			X
Cladocera			X
<u>Peltodytes</u> sp.	X		X
<u>Stagnicola</u> sp.			X
<u>Sialis</u> sp.			X
<u>Belostoma fluminea</u>			X
Coenagrionidae	X		X
<u>Dinuteus</u> sp.	X		
<u>Ambloplites rupestris</u>	X		
<u>Amblema costata</u>	X		
Comphidae	X		
Planariidae	X		
<u>Lethocerus americanus</u>	X		
INTOLERANT:			
<u>Leptophlebia</u> sp.	X		X
<u>Ectoparia nervosa</u>			X
Psychomyiidae	X		
<u>Potamanthus</u> sp.	X		

Table 6 (con't.)

<u>Ephemera</u> sp.	X
<u>Stenonema</u> sp.	X
<u>Limnephilidae</u>	X

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TOLERANT	3	2	5
FACULTATIVE	15	1	14
INTOLERANT	6	0	2



The richness of these streams is reflected in the size of benthic populations that occur. It is largely in the areas below outfalls, for a number of miles, that conditions are severe enough to limit the species. Where outfalls occur one after the other, the streams have little chance to recover.

There is no question that the benthic food supply would be sufficient to support good populations of fish provided the oxygen, temperature, and sediment conditions allow for healthy conditions (also non-toxic conditions must be assumed).

Fish data were available on seven of the streams. Extensive sampling and lists had been accomplished on four of the streams. Table 7 lists the common and scientific names of fishes occurring in southeast Michigan streams. Abbreviations listed are used in subsequent tables. It is notable that the walleye is not listed for any of the Michigan streams, but was reported in the Maumee in Ohio.

Data for the Raisin River are given in Table 8. There is a general gradient diagonally down and across the table indicating more game species in the headwater stations and more rough fish species in the lower reaches. Important species in the upper sections are largemouth bass and black crappie both of which apparently occur in lakes at the headwaters. At the lower end of the Raisin, carp, hog suckers, and goldfish are common. It should be noted also that smallmouth bass occur in the middle and upper-lower reaches.

All of the fish sampling points for the Huron River (Table 9) were located in the upper half of the stream (Figure 8), thus little information is available on the lower part. The notable fishes of the upper Huron are brown trout and white suckers, both species found commonly in cool waters. The lower most station sampled was populated heavily by carp and redhorse although smallmouth bass also were present.

Several stations were sampled in both the main branch of the Clinton (Table 10) and the north branch of the Clinton (Table 11).

Table 7. Common and scientific names of fish noted in study rivers.

Common Name	Scientific Name	Abbreviation Used in Tables
GAME FISH:		
Largemouth bass	<u>Micropterus</u> <u>salmoides</u>	LMB
Smallmouth bass	<u>Micropterus</u> <u>dolomieu</u>	SMB
Green sunfish	<u>Lepomis cyanellus</u>	GS
Bluegill	<u>Lepomis</u> <u>macrochirus</u>	BG
Rock bass	<u>Ambloplites</u> <u>rupestris</u>	RB
Pumpkinseed	<u>Lepomis gibbosus</u>	PS
Redear sunfish	<u>Lepomis</u> <u>microlophus</u>	RS
Longear sunfish	<u>Lepomis megalotis</u>	LS
Yellow perch	<u>Perca flavescens</u>	YP
Black crappie	<u>Poxomis</u> <u>migromaculatus</u>	BC
White bass	<u>Roccus chrysops</u>	WB
Channel catfish	<u>Ictalurus</u> <u>punctatus</u>	CC
Northern pike	<u>Esox lucius</u>	NP
Brown trout	<u>Salmo trutta</u>	BNT
Brook trout	<u>Salvelinus</u> <u>fontinalis</u>	BKT
Pickering	<u>Esox spp.</u>	P
Walleye	<u>Stizostedion</u> <u>vitreum</u>	W
NON-GAME FISH:		
Black bullhead	<u>Ictalurus melas</u>	BB
Carp	<u>Cyprinus carpio</u>	C
Goldfish	<u>Carrassius auratus</u>	G
Yellow bullhead	<u>Ictalurus natalis</u>	YB
Common shiner	<u>Notropis cornutus</u>	CS
Logperch	<u>Percina caprodes</u>	LP
Stoneroller	<u>Campostoma anomalum</u>	SR
Golden shiner	<u>Notemigonus</u> <u>crysoleucas</u>	GSH
White sucker	<u>Catostomus commersoni</u>	WS
Central mudminnow	<u>Umbra limi</u>	CM

Table 7 (con't.)

Greenside darter	<u>Etheostoma</u>	
	<u>blennioides</u>	GSD
Hog sucker	<u>Hypentelium nigricans</u>	HS
Creek chub	<u>Semotilus atromaculatus</u>	CRC
Madtom	<u>Noturus sp.</u>	MT
Chestnut lamprey	<u>Ichthyomyzon castaneus</u>	CL
Gizzard Shad	<u>Dorosoma cepedianum</u>	CZS
Sheepshead	<u>Apoldinotus grunniens</u>	SS
Emerald shiner	<u>Notropis atherinoides</u>	ES
Redhorse	<u>Moxostoma sp.</u>	RH
Bluntnose minnow	<u>Pimephales notatus</u>	BM
Bowfin	<u>Amia calva</u>	DF
Stonecat	<u>Noturus flavus</u>	SC
Lake chubsucker	<u>Erimyzon succetta</u>	LC

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Table 8. Fish numbers in Raisin River. Station numbers refer to locations on Figure 8. Fish sampled with net at first two stations, while shocking techniques were used at other stations. Table 7 lists abbreviations for fish.

Station	Location	Date	Game Fish										Non-Game Fish									
			LMB	RB	PS	BG	BC	NP	LS	YP	GS	SMB	BB	LC	C	BF	WS	SC	HS	RH	GF	
1	Goose Lk.	8-71	2	2	5	2	6							8	1							
	Norwall Lake	"	3		12	213	153	4						2		9	2	2				
2	Waldron Rd.	"			12									1								
3	County Line Rd.	"		16		6			1													
4	Jefferson Road	"		1						1						1	29	1				
5	Brooklyn Road	"		16	4	5		1	3	1						3	11	5				
6	Mill Road (Dam)	"	3	16	1	85	1					10	3			1	2	1				
7	US 12																					
	Clin-ton Rd.	"	1	4	1		3		1			3				2	3		9	18		
8	Comfort Rd.	"		3	1																	
8A	Deerfield Road	"										3	1	1				1	4			
9	US 223																					
	Cadmus Road	"						2								1						

Table 8 (con't.)

10	Below Deer- field	1	1	2	8	1
11	WTP M-50	1	1	2	8	1
12	Dundee Raisinv.	9		18	3	1
13	Rd.	31	1	66	10	4
	Waterloo Dam					
14	Monroe I-75	1	7	8	2	5
	Monroe	1	1	9		7
						9



Table 9. Fish populations in the Huron River. Station numbers refer to locations on Figure 8. Abbreviations used are identified in Table 7.

Station	Location	Date	RB	BNT	BC	BG	NP	P	PS	YP	SMB	LMB	C	WS	CRC	HS	RH	BM	BF	YB
15	T3N, R8E Oakland Co.	3-68								2					9					
16	T3N, R8#, S3 Oakland Co. East Br. south of Crosby Lake Road	8-71		45					1					10						
17	T3N, R8E, S14, 15 Oakland Co. Teggerdine Road	8-71		18				1												
18	T2N, R7, 8E Oakland Co.	10-71	92	3	58	19	3	3						8						
19	Washtenaw Co. Maple Road	5-72	3		40		1			1	7	1	150	17		2	25	8	1	1

Table 10. Fish populations in the Clinton River. Station numbers refer to locations on Figure 8.  
Abbreviations used are identified in Table 7.

Station	Location	Date	Game Fish							Non-Game Fish						
			LMB	PS	RB	BG	RS	BC	YB	BB	CS	LP	C	SR	WS	GS
20	T4N, R9E See Van Norm. Dam to Rt. 10	5-70	2	5	5	4			2	1	1	2				
21	T3N, R9E, S33 Cooley Lk Road	6-67			+	+	+		+			+		+		
22	T3N, R10E S25 Squirrel Road	6-71									1					1
23	T3N, R10E S25 Hamlin Rd	6-71						5			1				1	1
24	T3N, R11E S16 Avon Rd	6-71													11	

Table 10 (con't.)

25	T2N, R3E S19 Garfield Road	6-71	3	1
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Stations 22 through 25 are located below Pontiac Waste Treatment Plant  
+ indicates presence.

Table 11. Fish populations in the North Branch of the Clinton River. Station numbers refer to locations on Figure 8. Abbreviations used are identified in Table 7.

Station	Location	Date	Game Fish					Non-Game Fish									
			GS	RB	SMB	NP	PS	BKT	CRC	MM	GSD	C	HS	WS	CS	SR	MT
26	T5N, R12E S3 above Kidder Road	5-71	1	1						15	4	1					
27	T2N, R14E S17 south of McKay Rd	5-71						10	2								
28	T2N, R14E S17 above 34 Mile Road	5-71		3						Many		20		5	Many		
29	T2N, R14E S17 at 30 Mile Road Bridge	5-71		7	1		3							6	Many	10	4

Table 11 (con't.)

30	T4, 5N R12, 13E from 26 Mile to 30 Mile Road	5-71	Few	Few	Few	Many	Many
----	---	------	-----	-----	-----	------	------



Above the city of Pontiac on the mainstream there are populations of largemouth bass and other sunfish. These game fish disappear below the wastewater treatment plant of Pontiac. Only a few non-game fish are found at the stations below Pontiac. The north branch of the Clinton is a much better stream particularly in the headwaters. Brook trout located at station 27 is evidence of good quality, cold water. The water probably warms and loses quality as it proceeds downstream where populations of suckers and finally carp replace most of the game species.

The Rouge River contains many species of fish as shown in Table 12; however, it should be noted that most game fish are rare and a few common, while the coarse fish are common and abundant (rare in a few instances). It is probable again that the brown trout and other game fish are more abundant in the headwaters and the rough fish more abundant downstream. The report indicated that in most instances the sport fish were few in number and small in size. The general appraisal of the stream for sport fishing was poor. Many stations had extensive mats of bluegreen algae on the bottom and downstream (Station 32, Figure 8) the bottom is oily and often anaerobic. Sparse data were available on the remaining streams. The following are quotes from district biologists in the respective areas.

*"The Belle River in the middle (Station 31, Figure 8) reaches supports populations of smallmouth bass, northern pike and rockbass. Common shiners were the only non-game fish sampled."*

*"Game fish such as walleye and white bass exist in the upper part of the Detroit River (Station 33, Figure 8) while downstream the river and fishery deteriorates. Largely carp and goldfish are found on the Michigan side."*

*"The Maumee River is improving in quality. Smallmouth bass are stocked near Toledo. Game fish in the river include yellow perch, channel catfish, rockbass, walleye, and white bass; while non-game fish are bullheads, many minnow species, quillback, darters, trout-perch, carp and sheepshead. Game fish are found even in the Toledo section of the river."*

No data were available on the Black, the Pine or the St. Clair; however, it is expected that these streams would probably

Table 12. Fish collected in Rouge River, Michigan (A = Abundant, C = Common, R = Rare, - = Absent). Stations were throughout the length of the stream.

Common Name	Middle Br.	Lower Br.	Upper Br.	Main Br.
GAME FISH:				
Brown Trout	R	-	-	-
Northern Pike	R	-	-	-
Largemouth Bass	R	-	R	R
Green Sunfish	C	C	-	C
Rock Bass	-	-	-	R
Black Crappie	C	-	-	R
Bluegill	C	-	-	C
Pumpkinseed Sunfish	C	C	C	C
Brown Bullhead	C	-	R	-
Longear Sunfish	-	-	-	R
COARSE FISH:				
White Sucker	A	A	A	A
Carp	C	C	R	R
Goldfish	C	R	R	C
FORAGE FISH:				
Creek Chub	A	A	A	A
Blacknose Dace	C	-	C	C
Northern Redbelly Dace	-	-	R	-
Redside Dace	R	-	R	-
Common Shiner	C	A	C	C
Golden Shiner	R	-	R	-
Fathead Minnow	A	A	C	A
Central Mudminnow	C	R	C	C
Johnny Darter	R	R	R	-
Rainbow Darter	R	R	-	-
Mottled Sculpin	R	R	R	R
Brook Stickleback	R	R	R	R
Stonerollers	C	C	C	C

have the better game fish species such as trout in the Black and Pine and smallmouth bass and walleye in the St. Clair.

The fish data somewhat modify impressions given by the water chemistry in that the Maumee appears better than the chemistry indicates, the upper Raisin is in fair condition but the lower part is indeed poor. The Rouge River has less game fish than the water chemistry would indicate. The streams are ranked in Table 13 on an overall ecological basis from poor to good.

Table 13. Subjective display of the relative ecological condition of southeastern Michigan streams and the Maumee of Ohio.

Stream	Poor	Fair	Good
St. Clair			_____
Black			_____
Pine			_____
Belle			_____
Clinton			
North			_____
Main		_____	
Rouge		_____	
Detroit		_____	
Huron			_____
Raisin	_____		
Maumee		_____	

### SECTION III - BASE CONDITIONS, PHYSICAL- CHEMICAL STRUCTURE

This section of the report will characterize the ecologically significant physical and chemical state variables in the southeastern Michigan area. The chemical factors discussed will include both those that promote biological productivity (nutrients) and those that act as toxic agents in the ecosystem (pesticides and certain heavy metals). The principal physical factors to be discussed include temperature and existing hydrologic regimes in watersheds. The identification and quantification of these chemical and physical factors will provide the necessary base for calculating the final abiotic nature of southeast Michigan study area after adopting a particular wastewater management scheme.

#### Pesticides and Hazardous Organic Compounds

Synthetic organic chemicals are used extensively in nearly every facet of modern society. A considerable variety of compounds are used in industrial and manufacturing processes, incorporated in various consumer products and, in the case of organic pesticides, they are applied over broad areas of both urban and agriculture land.

The majority of synthetic organic chemicals, including those considered highly toxic, are readily degradable to elementary materials which re-enter the chemical cycles of the biosphere. They pose no long-term hazard if applied or released in sufficiently small quantities to restrict the area or volume of contamination. The readily degradable organic compounds include many of the pesticides which are introduced deliberately into the environment to control plant or animal species adverse to human interests. The chemical compounds of ecological concern are those which enter the environment as waste materials in sufficient quantities to extend the intended area of degradation, or those which are sufficiently stable to resist degradation long enough to damage



biological communities. In addition, these persistent compounds may be accumulated by organisms directly from the environment or be concentrated in higher organisms including man by transfer through the food web.

The following discussion will provide an assessment of pesticides and other hazardous alien chemicals that may enter into or otherwise affect biological communities including man. Attention is focused primarily on the southeastern region of Michigan that would be influenced by the various wastewater management alternatives. The lack of detailed comprehensive records or data on pesticides or other hazardous chemicals for this region, however, requires some generalizations from other regions of the country.

Information is provided on major groups of compounds regarding their relative toxicity, factors influencing their distribution in the environment and potential hazards to human health or the integrity of ecosystems.

#### *Pesticide Use in Southeastern Michigan:*

The southeastern region of Michigan includes considerable agricultural land of important economic value. The major field crops of the region are corn, hay, wheat, other small grains and soybeans. Commercially-grown vegetables and orchards represent a small but important percentage of the harvested acreage. Detailed records of agricultural pesticide use for this region are not available but the acreage treated for various purposes in 1969 is given in Table 14. Herbicides to control weeds and grass in croplands constitutes the major use and insecticides applied to hay and other field crops is the second most widely used treatment. Insecticides and fungicides are also used extensively on vegetable and orchard crops. Other important agricultural uses include fly control around livestock, rodent control, seed treatment and fumigation of stored crop products.

The most widely used herbicides and insecticides for treatment of field crops is given in Table 15. Although these data apply to the state of Michigan in general, they are probably reasonably

Table 14. Acreage of agricultural land treated with sprays, dusts, fumigants in 1969 for various control purposes.<sup>a</sup>

County	Insects on Hay Crops	Insects on Other Crops	Nematodes In Crops	Disease in Crop or Orchard	Weeds or Grass in Crop	Weeds in Brush or Pasture
St. Clair	1,326	6,313	141	476	25,002	56
Macomb	565	4,618	5,443	2,430	13,753	109
Oakland	213	2,476		1,148	7,307	101
Livingston	2,617	3,974	141	734	21,059	318
Washtenaw	5,023	9,201	39	1,212	38,986	266
Wayne	215	2,650	62	387	8,535	2
Monroe	1,234	15,767	377	3,900	6,542	71
Lenawee	4,424	27,450	99	1,995	103,613	180

<sup>a</sup>U.S. Bureau of the Census, Census of Agriculture, 1969. Vol. 1 Area Reports. Pt. 13. Michigan, Section 2 Co. Data, U.S. Government Printing Office, Washington, D. C. 1972.

Table 15. Field crops treated with pesticide - estimated acres treated in 1969-1971.

Crop and Pesticide	Estimated Acres Treated (Thousands)		
	1969	1970	1971
INSECTICIDES			
Corn, preemergence			
Aldrin	128	155	166
Bux	----	----	14
Diazinon	36	21	38
Heptachlor	12	15	31
Small grains			
Carbaryl	108	149	172
Malathion	33	29	21
Hay			
Azinhosmethyl	49	31	39
Carbaryl	32	57	48
Diazinon & methoxychlor	34	44	25
Malathion	30	26	17
Malathion & methoxychlor	54	48	30
-----			
HERBICIDES			
Corn, preemergence			
Atrazine	739	740	872
Atrazine & alachlor	----	26	227
Atrazine & propachlor	29	29	----
Atrazine & linuron	42	28	41
2,4-D ester	46	41	33
Corn, postemergence			
Atrazine	180	273	326
Atrazine with oil	259	311	398
2,4-D amine	124	85	112
2,4-D ester	17	73	77

Table 15 (con't.)

Soybeans

Alachlor	11	31	18
Alachlor & linuron	----	21	54
Amiben	104	89	95
Linuron	142	152	173
Trifluralin	----	----	11

Small grains

MCPA	----	87	106
2,4-D amine	----	108	107
2,4-D ester	----	69	70

Dry beans

Amiben	5/	14	21
Amiben & CDAA	5/	32	19
EPTC	5/	345	370
Trifluralin	5/	44	48

---

accurate for southeastern portion of the state as well. Atrazine used alone or in mixture with other herbicides is the most widely used pesticide in Michigan. In 1971 atrazine was applied to over nine-tenths of the corn acreage before plant emergence and to three-fourths of the acreage as a postemergence treatment. Linuron was used on nearly one-half of the treated soybean acreage, and 2,4-D and MCPA were the primary herbicides used on wheat and small grain crops. Aldrin, an organochlorine insecticide, was used on three-fifths of the corn crops treated in Michigan. Heptachlor was the only other chlorinated hydrocarbon insecticide used on sizeable acreages of field crops. Approximately 200,000 acres of field crops were treated with these or other chlorinated hydrocarbon insecticides in 1971 (Anon., 1971c).

A wide variety of pesticides are recommended for treatment of vegetable and orchard crop lands. Many of these pesticides are applied in only limited quantity or to very limited acreage for specific purposes. Others are used for a variety of crops and are also found in pesticide formulations available for use in home gardens, and for treatment of ornamental trees, shrubs, and lawns. Some of the more important insecticides in this group include chlordane, methoxychlor, dieldrin, carbaryl, parathion, malathion, diazinon, guthion, disulfoton, toxaphene, endosulfan, and captan. Fungicides that are frequently used in agriculture include zineb, maneb, and polyram. Fumigants that are recommended for seed bed sterilization include methyl bromide, chloropicrin methyl isothiocyanate, sodium methyl dithiocarbamate, and dichloropropene (Anon., 1972e).

An important use of pesticides in southeastern Michigan has been for the control of the Japanese Beetle. From 1932, when it was first discovered in the Detroit area to the present time, annual applications of pesticides have been made to control the spread of this insect. Dieldrin and chlordane have been applied to soils in specific regions where beetle infestations were indicated by trapping surveys. Since 1968, new label restrictions



have limited the application of these pesticides in rural areas. Table 16 lists the applications of chlordane and dieldrin in southeastern Michigan from 1968 through 1971. The application rates for this purpose were heavy to insure prolonged insecticidal activity in the soil for several years.

Pesticides have and will continue to be used in urban areas to control insects of public health importance, nuisance insects, pests on ornamental and shade trees, lawns and for weed control on industrial sites, and right-of-ways. A general shift away from the more persistent chlorinated pesticides is evident in the urban areas as well as on agricultural land. Because of the lack of exposed soil surfaces where movement may be restricted due to adsorption, pesticides applied in urban areas are particularly subject to run-off to surface waters via storm sewers.

Other sources of pesticide contamination include wastes from pesticide manufacturing and formulation industries. Waste treatment by pesticide manufacturers differs according to location and types of compounds produced. Some common forms of treatment or waste disposal include deep well injection, or burial in land fills, incineration, municipal sewage treatment plants, biological treatment in sewage lagoons, and stabilization ponds, trickling filters, physical filtration and sedimentation and chemical treatment. Each method has limitations and none are totally efficient. Burial or disposal by injection below ground has not been carefully monitored for possible entry in ground water, although little is expected. Biological treatment will not suffice for some compounds and occasional slugs of toxic materials can actually upset these processes. Pesticides entering municipal sewage treatment facilities often pass through without degradation. Thus, concentrated pesticide levels in the sediments nearby outfalls of some cities have been traced to wastes from manufacturing plants (Anon., 1972f).

While it appears that suitable methods for treatment of pesticide manufacturing waste are available, very close control within the plant is apparently necessary and combinations of

Table 16. Record of acres treated to control Japanese Beetle infestations in selected counties of southeastern Michigan 1968-1971. (Data from Michigan Department of Agriculture; Plant Industry Division).

Year	County	Acres Treated	Pesticide	Formulation
1968	Monroe	50	Chlordane	Granular (10 lb/acre)
	Lenawee	45	Chlordane	Granular (10 lb/acre)
	Oakland	4	Chlordane	Granular (10 lb/acre)
		24	Dieldrin	Granular (2 lb/acre)
	Macomb	758	Chlordane	Granular (10 lb/acre)
		932	Dieldrin	Granular (2 lb/acre)
	Wayne	24	Dieldrin	Granular (2 lb/acre)
		8	Chlordane	Granular (1 lb/acre)
1969	Macomb	1136	Dieldrin	Granular (2 lb/acre)
		1214	Chlordane	Granular (10 lb/acre)
	Oakland	16	Chlordane	Granular (10 lb/acre)
	Monroe	8	Chlordane	Granular (10 lb/acre)
1970	Lenawee	204	Chlordane	Granular (5 lb/acre)
	Washtenaw	532	Chlordane	Granular (5 lb/acre)
	Macomb	600	Chlordane	Granular (5 lb/acre)
	Monroe	180	Chlordane	Granular (5 lb/acre)
	Oakland	36	Chlordane	Granular (5 lb/acre)
	St. Clair	60	Chlordane	Granular (5 lb/acre)
	Wayne	912	Chlordane	Granular (5 lb/acre)
1971	Washtenaw	26	Chlordane	Granular (5 lb/acre)
	Macomb	310	Chlordane	Granular (5 lb/acre)
	Monroe	143	Chlordane	Granular (5 lb/acre)
	Oakland	207	Chlordane	Granular (5 lb/acre)
	Wayne	1870	Chlordane	Granular (5 lb/acre)

several methods may be required to achieve acceptable effluents for some types of compounds (Kennedy, et al., 1972; Shuman, et al., 1972; Stojanovic, et al., 1972).

#### Environmental Contamination by Pesticides:

Pesticide contamination of the environment has been a subject of extensive investigation in recent years. In addition to a large number of individual publications in scientific journals, several major reviews have been published. The Report of the Secretary's Commission on Pesticides and their Relationship to Environmental Health (U.S.D.H.E.W., 1969) is a particularly useful review of major problems. One study that is especially pertinent to environmental concerns in Michigan is the Evaluation of DDT and Dieldrin in Lake Michigan by the Lake Michigan Interstate Pesticides Committee of the Lake Michigan Enforcement Conference (Anon., 1972d).

The fate of a pesticide in the environment is dependent upon its specific chemical properties; the formulation, the methods of application and the meteorological and edaphic conditions in the area of application. The dissemination of pesticides in the environment can occur through the atmosphere by drift from application, by volatilization from soil or water surfaces and in dust particles transported by high intensity winds. Pesticides may be transported in the atmosphere for great distance before fallout in precipitation or on particulate matter. Although this route of contamination is offered as an explanation for wide distribution of some pesticides in the environment, there are probably very few cases where substantial contamination has occurred except within a few miles or less of the application site (Weibel, et al., 1966).

Soil is a major reservoir of pesticide residues particularly from agricultural applications. Soil type, moisture content and temperature are all important factors determining the adsorption of pesticides on soil particles. The major mechanisms for retention of pesticides in soils involve their interaction with mineral

and/or organic colloids. Loss of pesticide activity in soils may be due to its disappearance through degradation or movement out of the soil, but it may simply indicate isolation due to physical interaction with soil materials. Moisture content can influence the relative importance of different mechanisms of adsorption or inactivation. Adsorption of organic molecules by clay minerals is promoted by dry soil conditions because of reduced competition for water molecules. An increased moisture content tends to displace them from mineral sites and thus increase their association with organic surfaces that are non-competitive with water. Pesticides tend to be adsorbed to a greater extent at cooler temperatures and physical adsorption can be reversed by increases in temperature (Wolcott, 1970).

Some degradation of pesticides occurs in soils due to activity of soil organisms, and interaction with organic and mineral matter including water. However, translocation into plants can also account for significant removal from the soil and into animal food chains. Root crops in particular may accumulate residues that are hazardous to consumers. Soil type, moisture and other conditions also influence this avenue of pesticide movement. This hazard is reduced by careful selection of both pesticide and crop types (Harris, 1970).

The major route of pesticide removal from soil is by precipitation or irrigation with subsequent run-off. Even pesticides that are strongly adsorbed to soil particles may be physically transported to aquatic systems. Movement of pesticides in soil is largely a function of solubility of the particular pesticide. Although considerable lateral movement may occur, much less pesticide is transported vertically in soil percolation. Thus, contamination of deep soil layers or ground water is not likely except in soils very low in organic matter (Lichtenstein, 1970). Highly contaminated soils resulting from heavy or repeated pesticide application may contribute residues to run-off water for several years. Additional losses may occur in soil transported by wind storms.



Water appears to be the eventual depository for pesticide residues that are lost from their point of application. Residues from run-off in urban and agricultural lands, sewage, industrial wastes, atmospheric fallout or careless dumping eventually enter water courses. In streams, much of the residue burden is carried downstream in conjunction with suspended sediments. If pesticide application in a watershed is ceased, a stream will eventually be decontaminated, but residues will be deposited in downstream catch basins, e.g. reservoirs, lakes and estuaries. The time required for degradation of pesticides in water is, of course, dependent upon the specific chemical and the chemical and physical properties of the water.

In lakes or ponds, pesticide residues may enter aquatic animals by both direct absorption and by uptake in the food chain. Not all pesticides will behave in the same manner and accumulation of some pesticides may be of only very limited duration. Persistent pesticide residues may also be cycled by water movements in the lake and subsequently translocated to aquatic organisms.

#### Organochlorine Insecticides:

The organochlorine insecticides have been widely used in pest control programs since the first applications of DDT in the 1940's. Their effective, residual activity for up to several years after application has promoted their use in treatment of soils, structural pest control, disease-vector control, and for household pest control.

The organochlorine insecticides typically have very low solubility in water, are highly lipophilic, and most are toxic to a wide range of animal species. They are very stable in the environment, resisting both chemical and biological degradation. Some soils and water sediments in areas receiving repeated applications have developed high concentrations of these compounds. Thus, even where application has ceased, continued leaching or cycling in water has caused environmental contamination.



All organochlorine insecticides are subject to both metabolic and non-metabolic degradation but at vastly different rates depending upon the compound and environmental conditions. Several of the organochlorine insecticides degrade rather rapidly to products that are highly persistent and biologically active. As examples, aldrin is epoxidized to dieldrin both in the soil and through animal metabolism, and heptachlor forms heptachlor epoxide by the same mechanism. These epoxides are both more toxic and more persistent than the parent compound. Another example is the degradation of DDT to DDD and DDE. DDE is the most common pesticide residue found in the environment. Although DDE is less toxic than DDT, it interferes with calcium metabolism in birds, resulting in eggshell thinning and reduced reproductive success. Derr and Zabik (1972) have also reported that DDE causes reduced egg viability of Chironomus tentans, an aquatic insect.

The mode of action of the organochlorine pesticides has not been clearly established at the molecular or biochemical level. It seems to be clear that all are neurotoxins, but the loci of action within the nervous system apparently varies from compound to compound. Direct toxicity through acute dosage is of less concern than the possible effects from chronic exposure and large accumulations in body tissues of non-target species. Laboratory data also indicate that chronically-exposed animals may suffer increased susceptibility to disease, reduced thermal tolerance, and reduced growth rates.

The apolar nature of organochlorine insecticides restricts their dispersion in water but enhances their accumulation in lipids of biological systems. They are usually tightly bound to organic and clay soils, with little penetration into subsoil through water percolation.

*diphenyl aliphatic compounds (i.e. DDT, DDE, Methoxychlor) --*

The use of DDT in Michigan has been largely curtailed. Residues presently found in the environment are primarily from previous contaminations of from input from the atmosphere. Some

input is also probable on products imported to the region. Future monitoring should show a continuous decline in environmental levels of DDT although DDE residues may remain relatively high for a longer period.

Methoxychlor has replaced DDT for many applications. It has a much shorter persistence time in the environment and does not pose the same hazards for residue accumulation.

*chlorinated aryl hydrocarbons (i.e. Aldrin, Dieldrin, Endrin, Endosulfan, Toxaphene, Chlordane, Benzene Hexachloride) --*

Aldrin and dieldrin are important insecticides in Michigan, particularly for treatment of corn crops. Dieldrin is a frequent contaminant in aquatic systems, including fish tissues.

Although residues of chlordane and toxaphene are not frequently reported in environmental samples, this may be due in part to analytical difficulties. Few monitoring programs have made special attempts to determine these residues. The analysis of these compounds is also subject to interference with other industrial waste residues.

#### Organic Phosphate Insecticides:

A wide range of organic phosphate insecticides are available for both agricultural and domestic pest control. The phosphate insecticides in general do not have a prolonged residual action and thus do not present the same hazards of accumulation in the environment. Their use has generally expanded in recent years to avoid residue problems and to control pests that have developed resistance to the organochlorine compounds.

The organic phosphate insecticides act by inhibiting the enzyme cholinesterase which results in excessive neural activity at synapses. Eventual blockage of nerve impulse transmission results. In general, mammals including man are more susceptible to these compounds than to the organochlorine insecticides. Their use presents a greater immediate hazard to applicators or to wildlife inadvertently dosed during application. Applications which drift

to or cover water surfaces can be lethal to aquatic species and residues can be transported considerable distances.

Degradation to less toxic products in water is influenced by pH with breakdown occurring slowly under acidic conditions and accelerated hydrolysis under alkaline conditions. Microbial degradation, photodecomposition, adsorption to particulate material, and temperature all influence the fate of these compounds in aquatic and terrestrial systems.

Parathion, malathion, and diazinon are organic phosphates that have become important substitutes for the organochlorine insecticides. Parathion in particular is extremely toxic to man and poses a significant hazard to applicators. Their lack of long residual activity also requires more frequent application for some pest control operations.

#### Carbamate Insecticides:

This group of compounds is a relatively recent origin, but includes insecticides that are very important for agricultural and domestic programs. The carbamates like phosphates, act on the cholinesterase enzyme system. Carbamates are rapidly detoxified and eliminated by animals and do not result in persistent residue in the environment. Carbaryl (Sevin) is one of the most common carbamate insecticides for agricultural and domestic use.

#### Herbicides:

Herbicide use has continually expanded in recent years with projected increases for the future. Many compounds are available with the largest quantity used for agricultural purposes. A second major use is in and near municipalities for maintenance of rights-of-way, industrial areas and for home gardens and lawns. Increasing amounts are also being used in pond, lakes, and waterways for control of aquatic plant growth.

Atrazine is the most widely used herbicide in Michigan, particularly for pre-emergent and post-emergent weed control in corn cultivation. Although it has wide usage and is moderately

persistent in soils, it has a low toxicity to mammals, fish, birds, and insects. Some atrazine may move to surface waters by run-off from poorly managed land, but monitoring has not shown it to be an important general contaminant. Hall, *et al.* (1972) have presented the first evidence of atrazine in run-off from agricultural land to surface waters, showing average run-off from corn cropland of 2.4% of that applied. Some run-off to adjacent oat crops caused plant damage.

The use of 2,4-D for many herbicide treatments in both agricultural and domestic situations has at times resulted in contaminated water but has not caused long-standing residue problems.

#### Polychlorinated Biphenyls (PCB):

Polychlorinated biphenyls (PCB's) are synthetic organic chemicals that have had extensive commercial and industrial use for the past forty years. Their unique physical and chemical properties have promoted their use in paints, electrical transformers, condensers, non-flammable oils, adhesives, plasticizers, heat transfer systems, hydraulic fluids, caulking materials, printing inks and many other processes where heat resistance and non-flammability are important properties. A rather thorough analysis of information on PCB properties and the distribution of PCB in the environment has been reported in the report on Polychlorinated Biphenyls in the Environment (Anon., 1972c). Some of this information is summarized below.

PCB's are structurally similar to DDT and have many similar properties including resistance to chemical and biological degradation in the environment. PCB's are rapidly absorbed by organisms; stored in body lipids, and they are subject to biological magnification through transfer in the food chain. Although PCB's have been used for a long time, their presence in biological material was not generally recognized until the 1960's. Only limited information is available on their toxic or sublethal effects. Aquatic organisms may concentrate PCB's by a factor



of 10,000 to 50,000 fold over water concentrations. Concentrations of 5 to 20  $\mu\text{g/liter}$  (parts per billion) are toxic to some aquatic organisms after prolonged exposure.

In the United States, PCB's have been manufactured by a single producer, the Montsano Company and marketed under the name "Aroclor." The Aroclors may consist of chlorinated biphenyls, chlorinated terphenyls, or a mixture of these compounds. The various Aroclors are identified according to the degree of chlorination of the molecule with approximately 102 probable isomers. The PCB's most commonly identified in environmental samples (1248 and 1254) are chlorinated biphenyls having 48% and 54% chlorine respectively.

The total domestic sales of PCB's in the U.S. during the period 1957 through 1971 ranged between 15,000 and 36,000 tons per year. The total loss to the environment between 1930 and 1970 is estimated to be approximately 390,000 tons. Nearly 300,000 tons are believed to have been deposited in dumps and landfills, 60,000 tons to aquatic environments, and 30,000 tons to the atmosphere.

The entry of PCB to the environment are known to occur from the following sources:

1. open burning of incomplete incineration of domestic and industrial wastes at usual temperatures (incineration at 2000° F or higher will destroy PCB's).
2. vaporization from paints, coatings and plastics.
3. municipal and some industrial sewers (both treated and untreated wastes).
4. accidental spills or poor waste disposal practices.
5. dumping of sewage sludge, municipal and industrial wastes.
6. sewage sludges disposed on land.
7. transfer from surface coatings (paints) and packaging materials into food and feeds.

*movement in the environment --*

PCB contamination of both terrestrial and aquatic environments is greatest in the vicinity of urban centers and industrial



sites. Concentrations in wastewaters from domestic and some industrial sites range from a few parts per billion to several hundred parts per billion. In treated municipal sewage effluents concentrations range from approximately 0.01 to as high as 76 ppb.

There are data that suggest aerial transport of PCB to areas remote from industrial sites or human habitations. Residues have been found in remote arctic lakes as well as in northern wilderness areas of Canada. Relatively high concentrations in oceanic birds and fish of the North Sea also suggest wide environmental distribution. PCB's adsorbed on particulate material that are airborne may account for much of the wide distribution.

Like the organochlorine pesticides, the distribution in aquatic systems is probably a downstream movement in rivers where PCB's adsorbed on suspended sediments eventually contaminate lakes or other receiving basins. Some loss must occur through sedimentation but the water circulation in lakes, movement of sediments in shallow nearshore areas and translocation by uptake in algae, zooplankton and fish promotes the availability to organisms for prolonged periods.

Experimental data indicate that fish, algae and other aquatic organisms rapidly accumulate PCB's directly from the water. Fish concentrate Aroclors 1248 and 1254 by factors of 10,000 to 76,000 fold over water concentrations. Because of their affinity for animal and plant lipids and resistance to degradation, the PCB's tend to be concentrated in the food chain. However direct environmental uptake appears to be the most important source of residues in the lower trophic levels.

The toxicity of PCB's to animals including aquatic species is variable depending upon the degree of chlorination. The greatest ecological significance is related to sublethal effects resulting from long-term exposure. Chronic exposure of mammals usually results in pathological changes in the liver, but in birds, pericardial edema, kidney damage and reduced spleen were more common.

PCB's are toxic to fish at concentrations of a few parts per billion or higher. Chronic exposure to sub-lethal levels has caused reduced growth rates and skin lesions. Some aquatic invertebrates are killed by PCB concentrations of less than one part per billion. Definitive toxicological information is still lacking, particularly regarding effects on man.

#### *use and regulation --*

The Montsanto Company has voluntarily limited the distribution of PCB to "closed system" use. In addition identification of sources of contamination by monitoring programs may reduce losses to the environment. However PCB's or substitute chemicals will continue to be used.

#### *Pthalic Acid Esters:*

Pthalate esters are synthetic organic chemicals that are in wide use for various products in industry, in the home and for the medical sciences. Most pthalate esters are plasticizers that are used in the manufacture of plastics, particularly polyvinyl chloride (PVC). The pthalic acid ester most commonly used as a plasticizer is di-2-ethyl hexyl phthalate (DEHP). They are incorporated into plastic formulations (up to 40% of the final weight) but the esters are not chemically bonded to the polymer matrix. Thus the esters are interspersed between the PVC matrix and serve as lubricants for movement or stressing of the plastic. Although the esters have a low volatility and very low water solubility. They do migrate from the plastic thus causing contamination of the environment.

PVC plastics containing the pthalic acid esters are widely used in construction, in wall coverings, upholstery, appliances, automobile interiors, tubing and plastic bags, food wrapping and wearing apparel. Di-n-butyl pthalate has been used as an insect repellent and in pesticide formulations to retard volatilization. Other pthalate esters are also used in plastics and in cosmetics and industrial oils.

Recent evidence has shown phthalate esters are rather common environmental contaminants. Dialkyl phthalate residues have been found in various segments of the aquatic environment particularly in samples of water, sediment and aquatic organisms taken near industrial and urban centers. Concentrations of a few parts per billion to several hundred parts per billion have been detected in rivers in the United States. The source of these contaminants in streams has not been well documented but sewage effluents do contain measurable concentrations.

Phthalate esters are concentrated by aquatic organisms and can accumulate in higher trophic levels by transfer in the food chain. Some laboratory experiments indicate fish and crustaceans can concentrate phthalate esters by a factor of several thousand fold over water concentrations. Laboratory bioassays indicate the acute toxicity of DEHP to aquatic organisms is relatively low with LC-50 values ranging from approximately 700 to 6000 parts per billion for various warm water fish species and Daphnia magna, a freshwater crustacean. DEHP caused reproductive failure in Daphnia, and reduced the reproductive success of some fish species exposed under laboratory conditions.

The ecological significance of these compounds is uncertain. There is the potential hazard to aquatic life as shown by the limited experimental data, but potential hazards to man is largely unknown. Man may be exposed by direct contact with plastic products in food packaging, medical products or other products in the home, as well as from potential residues in fish products and in water. Until more is known of the possible subtle effects on man or other organisms and their potential mutagenic and teratogenic properties these compounds should be considered potentially hazardous.

A limited amount of data on phthalate esters in the effluent of sewage treatment plants in southeastern Michigan has been obtained from the Michigan Department of Natural Resources, Bureau of Water Management (Table 17). The concentrations vary widely between various waste treatment plants but even greater variation

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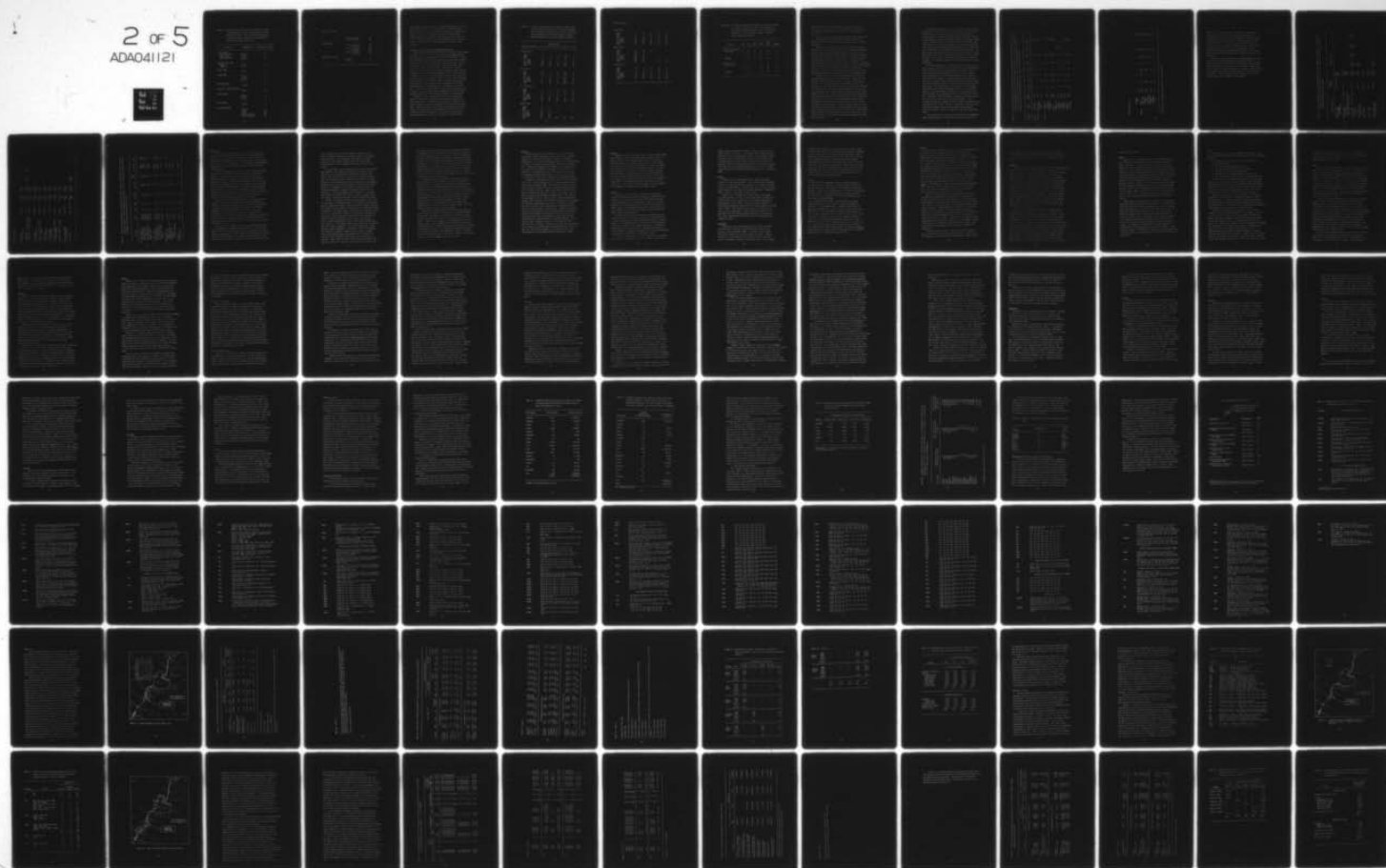


Table 17. Ethyl hexyl pthalates in final effluents of sewage treatment plants in selected southeastern Michigan communities, 1972. (Data from Michigan Department of Natural Resources, Bureau of Water Management, Water Quality Appraisal Section).

Location	Sampling Date	Concentration (ppb)
Pontiac WWTP		
East Boulevard	3-30-72	13
Auburn Road	3-30-72	4
East Boulevard	6-6-72	2
	6-7-72	1
Ypsilanti Twp. WWTP		
Number 1	4-3-72	4
Number 2	4-3-72	2
Monroe WWTP		
	3-3-72	12
Adrian WWTP		
	3-30-72	1
	6-12-72	3
	6-14-72	2
Ann Arbor WWTP		
	3-30-72	4
Wayne Co. (Wyandotte) WWTP		
	3-30-72	18
Trenton WWTP		
	3-30-72	1
	6-6-72	4
	6-7-72	4
Warren WWTP		
	3-30-72	5
Mt. Clemens WWTP		
	3-30-72	5
	6-7-72	42
	6-8-72	126
	7-26-72 Influent	10
	7-26-72 Effluent	10



Table 17 (con't.)

	8-9-72 Influent	19
	8-2-72 Effluent	11
Detroit WWTP		
	4-17-72 Influent	48
	Effluent	9
	6-1-72 Influent	121
	Effluent	23
	7-18-72 Influent	30
	Effluent	13
Detroit Pilot WWTP		
	6-1-72	4
	7-18-72	2

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occurs within individual plants on different sampling dates. These data suggest the occurrence of waste slugs containing phthalate esters which may occur within specific industrial or manufacturing plants. The data also indicate that waste treatment reduces phthalate ester concentrations in the final effluent by 50 to 75% of the influent concentrations. Some experimental data indicates phthalic acid esters can be reduced by 90% in activated sludge treatment.

#### Evidence of Contamination in Southeastern Michigan:

DDT, dieldrin (HEOD) and polychlorinated biphenyl (PCB) residues have been detected in each of seven major river systems sampled in southeastern Michigan (Table 18). The concentrations of total DDT (including DDD and DDE) is consistently low with very similar levels in each stream. The Huron River had particularly low concentrations in all sampling periods. Similarly dieldrin (HEOD) concentrations were never more than a few parts per trillion with especially low levels in the Huron and Detroit River samples. Although the series of samples from March, 1971 to June, 1972 show a progressive decline in concentrations, the data are insufficient to establish this as a definite trend. The use of DDT in Michigan has been largely curtailed and a decline in stream concentrations is an expected result of this action. Residues in the streams probably represent contaminated soil and sediments in the watershed. These concentrations are well below levels known to be toxic to aquatic life, but considerable accumulation in fish and other organisms may result from continuous exposure for long periods.

Freshwater clams or mussels have been used as biological monitors of pesticide concentrations in various streams in the Great Lakes region. Clams placed in streams for a specific time period accumulate residues in proportion to concentrations in the water. Mussels placed in two southeastern Michigan river basins during two time periods in 1970 accumulated very similar levels of DDT and dieldrin (Table 19). The total DDT concentrations averaged approximately 40 parts per billion. These levels are

Table 18. Pesticide and polychlorinated biphenyl (PCB) concentrations (parts per trillion) in river water samples from southeastern Michigan, 1971 and 1972. (Data from Michigan Department of Natural Resources, Bureau of Water Management, Water Quality Appraisal Section.)

Stream and Chemical	Sampling Dates				
	3-21-71	5-27-71	9-7-71	11-18-71	6-15-72
Clinton River					
HEOD	3	3	6	ND	1
DDE	6	7	1	3.0	2
TDE	1	2	7	<1	6
o, p-DDT	10	5	<1	<1	<1
p, p-DDT	20	18	2	4.6	8
Total DDT	38	52	10	6.6	8
PCBs	281	122	61	112	32
Black River					
HEOD	ND	1	.6	ND	<1
DDE	3	4	<1	2.0	2
TDE	<1	3	2	<1	<1
o, p-DDT	7	4	1	4.6	<1
p, p-DDT	13	11	2	4.6	<1
Total DDT	22	22	7	6.6	2
PCBs	3-	29	73	112	<10
St. Clair River					
HEOD	-	-	<1	<1	<1
DDE	3	5	7	<1	<1
TDE	3	7	<1	<1	<1
o, p-DDT	3	7	<1	<1	<1
p, p-DDT	11	14	2	<1	<1
Total DDT	18	26	14	<1	<1
PCBs	32	11	29	<10	<10
Raisin River					
HEOD	2	3	1		2
DDE	5	5			
TDE	3	6			
o, p-DDT	10	5			
p, p-DDT	20	18			
Total DDT	38	35			
PCBs	81	68	298	502	180

Table 18 (con't.)

## Huron River

HEOD	<1	<1	<1	<1	<1
DDE	5	7	1	<1	<1
TDE	6	5	6	2	<1
o, p-DDT	8	8	<1	<1	<1
p, p-DDT	17	20	3	<1	<1
Total DDT	36	40	11	2	<1
PCBs	22	20	39	19	<10

## River Rouge

HEOD	<1	ND	.6		<1
DDE	<11	9			
TDE	22	3			
o, p-DDT	19	5			
p, p-DDT	40	14			
Total DDT	122	32			
PCBs	959	67		327	120

## Detroit River

HEOD	ND	<1	<1	<1	<1
DDE	5	4	1	<1	<1
TDE	<1	3	7	<1	<1
o, p-DDT	8	5	<1	<1	<1
p, p-DDT	23	12	3	<1	<1
Total DDT	35	24	12	<1	<1
PCBs	16	34	53	36	<10

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Table 19. Pesticides in mussels held captive in selected streams in southeastern Michigan, 1970. Concentrations in parts per billion, wet weight. (Data from Michigan Department of Natural Resources, Bureau of Water Quality Appraisal Section).

Location	DDE	DDD	o, p-DDT	p, p-DDT	Total DDT	Dieldrin
Lake St. Clair Basin						
Salt River	6	15	3	11	35	3
	8	23	3	16	50	1
Average	7	19	3	14	43	2
Lake Erie Basin						
Otter Creek	6	16	2	6	30	<1
	8	27	3	10	48	<1
Average	7	22	3	8	39	<1



slightly lower than the average DDT concentrations found in tributary streams entering in the eastern side of Lake Michigan. The dieldrin concentrations were also low in the southeastern streams.

The data from water and mussel monitoring suggest that water concentrations are relatively low and probably result from general contamination of the watershed rather than point source discharges.

Data were not available on pesticide residues in municipal wastewater treatment plant effluents in the southeastern Michigan region. Some data from Wisconsin indicates relatively low to non-detectable levels of dieldrin and DDT in effluents from most Wisconsin municipalities. Some effluents in Wisconsin contained concentrations less than a part per billion of DDT and only a few had concentrations greater than 1 part per billion. Occasional samples indicated contaminations with lindane, aldrin and heptachlor epoxide. Similar results would be expected for wastewater effluents in Michigan. An exception would occur where pesticide manufacturing wastes were discharged to a waste treatment plant.

Pesticide residues in fish from Lake Erie and the major streams tributary to Lake Erie are indicative of both past and present contamination in the region. Reinert (1970) analyzed 13 species of fish from Lake Erie during 1967 and 1968. Concentrations of DDT and dieldrin were on the average only one-half to one-third the levels found in Lake Michigan fish. Only fish from Lake Superior had lower average residue concentrations than those from Lake Erie. Results from the National Pesticide Monitoring program (Henderson, 1971) show similar results indicating a relatively low level of DDT and dieldrin in Lake Erie fish. Considerable differences are observed between species of fish according to the fat content. Thus yellow perch which has a low percentage fat averaged about 0.87 ppm DDT while alewives, which have a higher percentage fat content, average about 1.59 ppb DDT (Reinert, 1970). The pesticide concentration in Lake Erie fish did not exceed Food and Drug Administration guidelines for either DDT or dieldrin.

The Michigan Department of Natural Resources, Bureau of Water Management has analyzed a variety of warm water fish species from the Raisin River at sampling stations from the headwaters in Lenawee County to the mouth of Monroe County (Table 20). Residue concentrations of dieldrin and PCB were considerably higher in fish taken at sampling sites in the lower reaches of the stream. DDT concentrations in fish were relatively low in the upper sampling sites in Lenawee and Washtenaw counties but general increases with some fluctuations occur through the remaining sampling sites. The DDT concentrations are similar to those observed in fish from other watersheds in the state. The dieldrin concentrations in the lower portions of the river are considerably higher than are observed elsewhere. The specific source of this contamination has not been identified.

The PCB concentrations in the Raisin River fish show a distinct increase below the city of Tecumseh and continued high levels below the confluence of the South Branch of the Raisin River. The relatively high concentrations of Aroclor 1242 in samples taken just below the city of Tecumseh suggest a point source of this PCB from an industry. The general increase of PCB concentrations in stations below the South Branch indicate a major input from the South Branch as well as sources in the lower reaches of the stream. PCB concentrations in the majority of fish samples were below the Food and Drug Administration limit of 5.0 mg/kg. However, these data indicate a significant contamination in the lower watershed and a significant input to the Lake Erie environment. Henderson (1971) found PCB concentrations in three species of fish taken in Lake Erie, near Erie, Pennsylvania. They averaged 1.94 ppm in freshwaterers drum, 2.33 ppm in yellow perch and 2.48 ppm in white sucker. By way of comparison, yellow perch and white sucker samples taken in Lake Michigan near Sheboygan, Wisconsin averaged 12.6 and 14.8 ppm PCB respectively (Henderson, 1971).

PCB concentrations found in the final effluents of municipal waste treatment plants in southeastern Michigan are listed in

Table 20. Chlorinated hydrocarbon insecticides and polychlorinated biphenyl (PCB) residues in fish from selected sampling sites on the Raisin River<sup>1</sup> in August, 1971. Concentrations expressed as parts per billion on wet weight basis. (Data from Michigan Department of Natural Resources, Bureau of Water Management, Water Quality Appraisal Section).

Location County	Fish Species	Fish Length	Dieldrin	DDE	DDD	O, p-DDT	P, p-DDT	Total DDT	PCB as 1242	PCB as 1254
Lenawee	Rock Bass	8.5	<1	2	<1	<1	<1	2		10
Washtenaw	Rock Bass	9.0	<1	8	1	<1	<1	9		58
	Smallmouth Bass	8.0	3	11	<1	<1	11	23		76
Lenawee	Smallmouth Bass	14.5	1	22	22	<1	17	62		13
	Rock Bass	7.8	1	9	13	2	12	36		30
	Smallmouth Bass	9.0	4	35	29	<1	23	88		20
	Carp	16.0	1	6	2	<1	16	25		50
	Carp	16.0	7	55	90	<1	26	171		308
	Smallmouth Bass	14.0	7	14	10	<1	13	37	4710	
	Northern Pike	20.0	3	4	2	<1	3	9	6450	
	Northern Pike	17.0	1	18	6	<1	7	31		1470
	Northern Pike	28.0	3	8	16	<1	10	34		1180
	Rock Bass	7.0	3	3	13	<1	15	34		130

Table 20 (con't.)

	Northern Pike	15.5	2	13	7	<1	6	27	590
Monroe	Carp	12.5	7	12	84	<1	32	128	420
	Rock Bass	7.5	3	6	12	<1	45	63	30
	Northern Pike	23.5	42	7	50	<1	33	90	310
	Rock Bass	7.5	17	4	9	<1	1	13	400
	Smallmouth Bass	15.0	15	7	25	<1	26	58	120
	Smallmouth Bass	11.5	39	14	46	<1	42	101	140

<sup>1</sup> Sampling sites numbered from upstream to downstream areas.

Table 21. These data indicate a substantial contamination of receiving waters with PCB from municipal waste effluents. In particular the Detroit and Adrian wastewater treatment effluents are quite high. A limited number of influent-effluent sample comparisons show PCB concentrations are reduced by one-half or more in the final effluent. An important quantity of PCB removed in waste treatment is found in the sludge (Anon., 1972c). Additional environmental contamination may occur if the sludge wastes are deposited where surface or ground waters may be contaminated.

Pesticide and PCB concentrations in water supplies for major cities in southeastern Michigan are presented in Table 22. In every instance the concentrations are well below the recommended limits for public drinking water. Only the water supplies from the St. Clair River and the Wyandotte water supply from the Detroit River indicate measurable concentrations over the entire period of sampling. PCB concentrations were consistently low or non-detected except for the Algonac water supply.



Table 21. PCB's in municipal wastewater treatment plants in southeastern Michigan. Concentrations in parts per trillion (ppt) as Aroclor 1254. All samples were 8 hour final effluent composites unless otherwise stated.

Location and Date	Receiving Water	Sampling Date
Brighton WWTP	Brighton Lake	10/5/71
Raw Sewage		810
Final Effluent		380
Detroit WWTP - 24 hr. composite	Detroit R	10/14/71
Raw Sewage		4600
Final Effluent Unchlorinated		2200
Grab Sample Chlorinated		3600
Detroit WWTP - 10 hr. composite	Detroit R	12-20-71
Influent		5800
Effluent		3000
Ann Arbor WWTP	Huron R	10-14-71
Final Effluent		140
Dexter WWTP	Huron R	10-14-71
		50
Milford WWTP	Huron R	10-14-71
		44
Mount Clemens WWTP	Clinton R	11-15-71
Final Effluent		1400
		3-30-72
		2900
		1-20-72
		3200
		920
		3-9-72
		5700
		2400
		4-17-72
		9800
		1800
		6-1-72
		17000
		880

Table 21 (con't.)

Warren WWTP Final Effluent	Clinton R	11-15-71 160	3-30-72 100		
Pontiac WWTP East Blvd. - Final Effluent	Clinton R	11-17-71 200	3-30-72 1300	6-6-72 150	6-7-72 160
Auburn Rd. - Final Effluent	Clinton R	11-17-71 68	3-30-72 610		
Monroe WWTP Final Effluent	Raisin R	11-15-71 600	3-30-72 330		
Wayne Co. (Wyandotte) WWTP Final Effluent	Detroit R	11-16-71 640	3-30-72 170		
Trenton WWTP Final Effluent	Detroit R	11-16-71 140	3-30-72 1100		
Port Huron WWTP Final Effluent	St. Clair R	11-15-71 280	3-30-72 520		
Ypsilanti Twnship. #1 WWTP Final Effluent	Huron R	11-16-71 120	4-2-72 <100		
Ypsilanti Twnship. #2 WWTP	Huron R	11-16-71 190	4-2-72 160		
Mount Clemens WWTP Final Effluent	Clinton R	6-7-72 4500	6-8-72 3100		
Adrian WWTP	River Raisin	3-30-72 1800	6-13-72 14,000	6-14-72 6900	

Table 22. Pesticide concentrations in interstate water intake samples, Spring 1971 (Concentrations in parts per trillion. Data from Michigan Department of Natural Resources, Bureau of Water Management, Water Quality Appraisal Section).

Location	Date	DDE	DDD	o, p-DDT	p, p-DDT	Total DDT	Dieldrin	PCB
St. Clair River								
Port Huron W.W.	3-11-71	<1	<1	2	14	17	0.001	<10
Marysville W.W.	3-11-72	<1	<1	2	10	13	N.D.	<10
St. Clair W.W.	3-11-71	2	4	12	20	38	0.001	<10
E. China Twp. W.W.	5-17-71	<1	<1	<1	2	2		<10
Marine City W.W.	3-11-71	2	3	19	25	49	0.001	
Algonac W.W.	4-21-71	<1	4	2	15	21	N.D.	28
Lake St. Clair								
Fairhaven W.W.	5-17-71	<1	<1	<1	<1	<1	<0.001	<10
New Baltimore W.W.	4-21-71	<1	<1	<1	<1	<1	N.D.	<10
Mt. Clemens W.W.	4-21-71	<1	<1	<1	6	7	<0.001	<10
Grosse Pte. Farms W.W.	4-21-71	<1	<1	<1	3	3	N.D.	<10
Detroit River								
Detroit - E. Jefferson Ave. W.W.	4-21-71	<1	<1	<1	<1	<1	N.D.	<10
Wyandotte W.W.	2-9-71	1	3	2	10	16	0.001	
Detroit - Allen Park W.W.	2-9-71	<1	<1	<1	<1	<1	N.D.	
Lake Erie								
Monroe W.W.	2-9-71	<1	5	<1	5	10	0.001	

### Heavy Metals

Until recently, the main source of heavy metals into the environment has been erosion and leaching from geological formations to streams and lakes by ground water runoff. Since the industrial revolution, however, increasing amounts of heavy metals have been lost to the environment as waste products from manufacturing processes or from the improper disposal of industrial and consumer products.

In urban and industrial areas heavy metal containing consumer products such as cosmetics, paints, chemicals or medicines are disposed of through the sanitary sewer systems. These heavy metal discharges are too small to be significant individually. However, altogether, potentially large quantities of these heavy metals are discharged to the watercourses that receive the effluent from sewage treatment plants. In addition, large quantities of heavy metals are lost to the environment through open refuse burning, automobile emissions, burning of fossil fuels, industrial manufacturing emissions, as well as through the smelting of ores to recover such metals as copper, lead and zinc.

Whenever a heavy metal containing ion or compound is introduced into an aquatic environment, it may undergo hydrolysis, chelation, complexation, precipitation or adsorption onto organic or inorganic particulate material in the water. Furthermore, depending on the original chemical state of the heavy metal introduced, it is highly probable that two or more of these reactions take place before its chemical equilibrium is attained. In natural waters so much partitioning of these materials onto particulate matter occurs that dissolved ionic heavy metal species are unlikely to exist.

Heavy metals are removed from the water column and deposited in the sediments by adsorption and precipitation. How the heavy metal species are further partitioned in the sediments is not known. However, they appear to be most commonly associated with the organic fraction of the sediments. Some heavy metals (vis mercury and copper) have been reported to be almost quantitatively partitioned

into the organic fraction of the sediments (Upchurch, 1972). Thus, the organic fractions of sediments appear to act as heavy metal reservoirs. As the physical, biological, and chemical conditions in the aquatic system change, these organically bound heavy metals can be released and recycled into the water column.

Toxic pollution can result when a heavy metal interferes with some metabolic life process of the biota in an aquatic system. Depending on the concentration of these toxic materials, the direct effects on the biota can vary from no apparent or measurable toxic symptoms to death. Indirectly, the effects of these pollutants can be responsible for: (1) elimination of an organism's primary food source, (2) subtle changes in the organism's normal behavior, (3) changes in the reproductive process as well as decreased hatchability of eggs, (4) reduce the ability of the organism to resist environmental stress conditions. Thus heavy metal pollution acting on the biota of an aquatic system can, to some degree, control the diversity of species within that system.

Chronic or subchronic toxicities of heavy metals on the biota are difficult to establish because of the chemical, biological and physical complexities within a natural aquatic ecosystem. Moreover, researchers have been unable to recognize specific sublethal effects as endpoints. For these reasons, most of the bioassay studies reported in the scientific literature are designed to measure acute toxicities. These acute toxicity data show a wide variation of reported lethal concentrations of heavy metals for bioassay tests of equal time duration. Most of these studies report only short term (i.e., less than one week) acute toxicities for selected species of aquatic organisms. The chemical parameters of these tests are widely divergent or uncontrolled. Thus, the findings are difficult to compare or apply to other studies. Usually these acute bioassay tests are performed under laboratory conditions in which completely ionized heavy metal species act as toxicants. Many chemical, physical and biological parameters are neglected because they cannot be controlled in this type of test.



Often these neglected parameters exhibit substantial antagonistic or synergistic effects and may drastically change the absolute toxicity of the heavy metal. Such factors include: temperature, pH, organic pollution loadings, hardness, suspended particulate loadings, alkalinity, heavy metal loadings, and dissolved oxygen concentrations. The relative toxicity of heavy metals to aquatic organisms varies quite widely with the species, its life stage, feeding habits, metabolic rate, and age as well as its exposure interval. Finally, that aquatic organisms are able to concentrate heavy metals biologically is an important factor in the overall evaluation of heavy metal toxicants.

A completely accurate prediction of the amounts of heavy metals which are toxic to aquatic organism is not possible at the present time. Therefore, toxicological data obtained from laboratory bioassay experiments must be cautiously applied to determine heavy metal concentrations that are "safe" for aquatic organisms in a natural water system. The "safe" concentration for each heavy metal can be more realistically determined by first establishing the 96-hour  $LC_{50}$  for the most sensitive aquatic organism of that system and then applying a "safety factor" ranging from 0.1 to 0.01 depending on the persistence and cumulative effect of the toxicant on aquatic animals. Even this approach, however, has its serious drawbacks.

Twenty-one heavy metals have been selected which appear to pose the greatest pollutional threat in the southeastern region of Michigan. Depending on concentrations, all of these heavy metals can exhibit toxic effects on the biota in an aquatic ecosystem. Eight of these elements vis chromium, manganese, iron, cobalt, copper, zinc, molybdenum and selenium are required in trace amounts for the normal nutrition of either plants or animals. In addition, there are data to indicate that vanadium may also be an essential element. The remaining 12 elements considered here are nickel, beryllium, antimony, tin, cadmium, lead, arsenic, mercury, titanium, aluminum, barium, and silver.

#### Aluminum:

Aluminum is used in such industries as: textile building construction, light engineering, paper, glass, office equipment, appliances, electrical apparatuses, aircraft, and pharmacy. Additionally, it is used as a packaging material, in the production of alloys, to manufacture printing inks, and for water purification. Aluminum in drinking water supplies is not considered to be a public health problem, at least when it is present as a result of the treatment processes. Moreover, the total amount of aluminum ingested by human beings as a result of their diet is estimated at between 10 and 100 mg per day (Browning, 1969).

Aluminum is concentrated in natural waters by the weathering of rocks in a range between 0.012 to 2.55 ppm (Durum and Haffty, 1961; Faber and Bryson, 1960). While selected streams in California have been reported to have an average aluminum concentration of 0.015 ppm (Silvey, 1967), the Sacramento River at Sacramento, California has had levels as high as 1.11 ppm. Fish are lethally affected by aluminum at minimum levels in the range of 0.1 ppm (Murdock, 1953). However, recent studies (Freeman and Everhart, 1971) indicate that dissolved concentrations of aluminum are extremely pH dependent; and levels in excess of 1.5 ppm in the water can cause drastic physiological and behavioral aberrations as well as death to rainbow trout. Freeman and Everhart (1971) also reported that since aluminum adsorbs onto suspended materials the rainbow trout showed toxic effects at lower concentrations than those of the dissolved forms where the safe concentration of either dissolved or suspended aluminum is well below 0.5 ppm. Aluminum is also one of the inorganic salts which are toxic to plants. Therefore, for water used continuously on all soils the tolerance limit of aluminum has been set at 1.0 ppm. For short term irrigation on fine textured soils, water containing up to 20 ppm aluminum may be used (Anon., 1968a).

#### Antimony:

Antimony is used in the manufacture of storage battery grids, pewter, printer's type, lead shot, lead electrodes, bearing metals, rubber, textiles, paint, glass, and percussion caps. Although antimony is a relative toxic element to human beings and the fatal dose varies with individuals, as little as 100 mg has been reported as a fatal dose (Browning, 1969). Few data are available with respect to its effect on aquatic organisms. The 96-hour  $TL_m$  of fathead minnows for several antimony salts has been estimated to range between 9 and 20 ppm in soft water and between 12 and 17 ppm in hard water (Tarzwell and Henderson, 1956; 1960). The maximum permissible concentrations of this element or its salts have not been established for either public water supplies or irrigational waters (Anon., 1968).

#### Arsenic:

Arsenic is used in industry to manufacture glassware, ceramics, pharmaceuticals, dyes, pesticides, leather, and other products. It is also used to increase the hardening and heat resistance properties of certain metals. Arsenic is not considered to be an essential element to human metabolism and is highly toxic to all animals which possess a central nervous system. However, it is present in extremely small quantities in human tissues (Browning, 1969).

Arsenic usually occurs in the natural environment in the form of arsenites of true metals or pyrites, and concentrations in excess of 10 ppb are rarely encountered in surface or ground waters. The maximum permissible concentration of this element allowed by the United States Public Health Service (USPHS) for drinking water standards is 50 ppb. However, because arsenic has been shown to accumulate in human tissue, the USPHS recommends limiting the concentration to 10 ppb (Anon., 1968).

Arsenic concentrations as low as 1.1 ppm have been reported to be toxic to fish (Anon., 1950; Anon., 1941). Lower forms of

aquatic organisms appear to be able to tolerate higher levels of arsenic with no ill effects. For example, mayfly nymphs are not harmed by arsenic concentrations of from 3 to 14 ppm while dragon and damselflies are unaffected by arsenic levels as high as 10 to 20 ppm (Rudolfs, et al., 1950). Because of its phytotoxic properties the level of arsenic has been set at 1.0 ppm in irrigation water which is sprayed continuously at one site. For short term irrigation water containing 10 ppm arsenic can be used (Anon., 1968).

#### Barium:

Barium is used to manufacture paint, cement, glassware, ceramics, alloys, and pesticides. Since barium levels in natural soils are in the range of 300 to 500 ppm, it is normally found in surface and ground waters. Although many barium salts are soluble, barium carbonate and sulfate are extremely insoluble. Therefore, barium levels are quite low in natural waters.

Barium is not considered an essential element for animal or plant nutrition. In fact, it is a known muscle stimulant and may have toxic effects on the heart, blood vessels and nervous system (Derby, et al., 1960). For this reason the USPHS set the maximum permissible concentration for drinking water at 1.0 ppm (Anon., 1968). While barium and its salts do not appear to be toxic to aquatic organisms in concentrations less than 10 to 50 ppm (McKee and Wolf, 1963), goldfish apparently can concentrate this element approximately 150 times from the surrounding water (Templeton, 1958).

#### Beryllium:

Beryllium is used in the manufacture of alloys, x-ray diffraction tubes, neon signs, fluorescent lamps, ceramics, and nuclear reactors (Browning, 1969). A relatively rare element, beryllium is highly toxic to both animals and plants. However, because it is not normally found in natural waters, the USPHS has not recommended a mandatory permissible limit for this element



in public water supplies or farmstead uses. A tolerance limit of 0.5 ppm has been set for irrigational waters because beryllium inhibits bean growth at this level (Ballard and Butler, 1966). Few data are available which describe the toxic effect of beryllium on aquatic organisms. Tarzwell and Henderson (1960) established that beryllium is much more toxic in soft water than hard water. The 96-hour  $TL_m$  values for fathead minnows and bluegills were found to be approximately 0.15 and 15 ppm in soft and hard water respectively. In addition, beryllium appears to be biologically concentrated in various animal tissues, and it is slowly excreted over a period of years.

#### Cobalt:

Cobalt and its compounds are used in the manufacture of alloys, in nuclear technology, as a blue pigment in the china and glass industry, and as binders in the tungsten carbide tool industry. Cobalt and its compounds have a relatively low toxicity for man, and it is an essential element in animal nutrition. Moreover, it does not appear to be retained in mammals to any great extent (Browning, 1969).

In natural waters cobalt is rarely present, and large enough concentrations to be lethal to aquatic organisms are unlikely, even in polluted waters. Cobalt concentrations of 1 ppm were not harmful to one year old tench, carp, rainbow trout, and char; or to the organisms that form their food: worms and insect larvae (Schweiger, 1957). However, as the cobalt ion concentration is increased to the range of 10 ppm, it exhibits a toxic effect on aquatic organisms (Jones, 1939; Ellis, 1937; Doudoroff and Katz, 1953). Finally, some aquatic organisms seem to be able to concentrate cobalt biologically from the water. For example, Morgan (1961) reported that the flagellate Ochromonas concentrated cobalt between 1070 to 1500 times while the diatom Navicula showed a concentration factor of 271.



### Cadmium:

Cadmium and its compounds are used in electroplating processes, in nuclear reactors as neutron absorbers, in pesticides, in storage batteries, in low melting alloys and as pigments in the manufacture of glassware and in some paints. Cadmium is a cumulative and extremely toxic poison in mammals, therefore, it can be an industrial hazard, food contaminant and freshwater pollutant. If cadmium has a physiological role it is unknown (Browning, 1969). The maximum permissible concentration of cadmium set by the USPHS Drinking Water Standards is 10 ppb because it is a cumulative poison which results in an insidious, progressive chronic poisoning of the organism that ingests it (Anon., 1968). Rats exposed for a lifetime to small doses in drinking water have exhibited arterial hypertension (Schroeder, 1964). Some evidence also suggests that cadmium is a cause of human hypertension (Schroeder, 1965). Moreover, very low cadmium concentrations in milk have been correlated with cardiovascular death rates among people in the United States (Pinkerton and Murthy, 1969).

While only minute quantities of cadmium are normally found in unpolluted natural waters, concentrations ranging from 1 to 10 ppb were detected in 42% of the samples in a recent survey of surface waters of the United States (Durum, *et al.*, 1971). Moreover, about 4% of the river samples had cadmium concentrations in excess of 10 ppb. Pickering and Henderson (1964) have reported cadmium 96-hour  $TL_m$  values as high as 73 ppm in hard waters. However, the concentrations of cadmium lethal to fish usually vary from 0.01 to 10 ppm and depend on the chemical and biological characteristics of the system. Cadmium acts synergistically with other heavy metals to increase its toxicity. For example, cadmium concentrations of 0.03 ppm in combination with 0.15 ppm of zinc from galvanized fish hatchery screens caused the mortality of salmon fry.

Ball (1967) reported that a lethal threshold of 10 ppb was not evident until about seven days after exposure. Safe levels of cadmium for both fathead minnows and bluegill sunfish in hard

water (200 ppm as  $\text{CaCO}_3$ ) appear to be between 30 and 60 ppb. In soft water (45 ppm as  $\text{CaCO}_3$ ) cadmium concentrations in excess of 5 to 10 ppb were found unsafe for several warm and cold water fishes.

#### Chromium:

Chromium is used in the metallurgical, electroplating, tanning, paint, pesticide, textile, and photographic industries. It is also widely used as a corrosion inhibitor in cooling and industrial waters. Chromium compounds are relatively non-toxic to human beings and animals because small amounts are rapidly eliminated from the human body. Moreover, while chromium has not been proven to be essential for human nutrition, it may be physiologically involved in the metabolism of fatty acid and insulin (Browning, 1969). The USPHS maximum concentration allowed in drinking water is 50 ppb for hexavalent chromium while no limit has been recommended for trivalent chromium. Chromium is a phytotoxic element with a recommended maximum concentration in irrigational waters set at 5.0 and 20.0 ppm respectively for long and short term use (Anon., 1968). Chromium rarely occurs naturally in rivers and streams, and its presence is almost completely associated with industrial or related processes.

The toxicity of chromium toward aquatic organisms varies widely. However, the lethal toxic effects of chromium toward fish are evident at concentrations between 0.1 and 1 ppm. These toxicity levels depend on the species, temperature, pH, valence of chromiums and water hardness. Hexavalent chromium has been shown to produce a cumulative toxicity in trout and salmon. Therefore, the safe concentration of hexavalent chromium in fish appears to be in the range of 0.05 ppm. Chromium has also been shown to inhibit the growth of some algae at concentration ranges between 0.1 and 1 ppm. The threshold lethal level for daphnia has been reported to be in the range of 0.016 to 0.7 ppm (Anon., 1950). Recent studies with radioactive chromium-51 indicate that the American oyster (Crassostrea virginica) biologically concentrates

chromium (Preston, 1971).

#### Copper:

Copper is used in the metallurgical, electroplating, pesticide, electrical, construction, textile, munitions, and photographic industries. It also serves as a catalyst in chemical manufacturing and as a mercaptan scavenger in petroleum refining operations. This element is essential to animal and plant metabolism, and its salts are relatively non-toxic to human beings and terrestrial animals in small amounts (Browning, 1969). Nonetheless, the USPHS maximum concentration allowed in drinking water is 1.0 ppm with the recommendation that it be completely eliminated whenever possible. Copper can be considered a phytotoxic element and its recommended maximum concentration in irrigational waters has been set at 0.2 and 5.0 ppm respectively for long and short term use (Anon., 1968). Copper salts occur in natural water only in trace amounts of up to about 0.05 ppm. Therefore, levels in excess of this value can normally be attributed to pollutional sources.

Copper has long been known to be extremely toxic to aquatic organisms, and many studies have been conducted to determine safe levels for their protection in natural waters. Based on studies with fathead minnows, a "safe" level for copper ions in soft and hard water range from 11 to 18 and 15 to 33 ppb respectively (Mount and Stephan, 1969). Many bioassays have shown that the toxicity of copper varies with changes in pH, hardness, temperature, alkalinity, and dissolved oxygen content of the water as well as with changes caused by biological variations in the test species. The formation of soluble and insoluble complexes of copper has also been demonstrated to alter the toxicity of copper in natural waters. The toxicity of mixtures containing copper and other heavy metals can be calculated as the total of all fractional toxicities.

### Iron:

The industrial uses of iron are numerous and varied. Iron can be introduced into the aquatic ecosystem by all of the following processes:

1. Leaching of soluble iron salts from soil and rocks
2. Acid mine drainage
3. Use of iron-containing ground waters
4. Iron pickling wastes from the steel industry
5. Corrosion of iron-containing materials
6. Burning of iron-containing materials such as coal
7. Use of iron-salts in tertiary waste treatment processes

Iron is an essential element in human and animal nutrition and has a very low toxicity. The taste threshold for iron salts is approximately 0.1 to 0.2 ppm and the USPHS drinking water mandatory limit is 0.3 ppm, primarily for esthetic or taste consideration and not for health reasons (Anon., 1968). Iron is not normally considered to be a problem in irrigation waters.

Many of the iron salts are water soluble, and in natural surface waters the ferrous ions are readily oxidized to ferric ions which react to produce an extremely insoluble precipitate of ferric hydroxide. Durfer and Becker (1964) reported that the concentrations of iron in typical surface and ground waters usually range between 0.00 to 0.72 ppm in the United States.

Few data are available on the toxicity of iron to fish although it has been studied frequently. For example, levels of iron ion as low as 0.2 ppm in soft water (Minkina, 1946) have killed fish whereas 100 ppm in hard water did not kill goldfish in four days (Ellis, 1937). If excessive levels of iron are present in an aquatic environment, the precipitated ferric hydroxide may be deposited on the fish's gill surfaces. Depending on the amount, the stress it induces on the fish may be fatal. In addition, heavy precipitation of ferric hydroxide can smother fish eggs and prevent their hatching. It appears that iron levels below 0.1 ppm should not constitute a hazard to fishes. However, as the concentrations increase beyond this level, they can be toxic

to selected fish species. Moreover, iron can be biologically concentrated by many species of aquatic organisms. Phytoplankton have possible biological concentration factors as high as 200,000 (McKee and Wolf, 1963).

Lead:

Lead is used in the glassware, ceramics, photography, textile, paint, metallurgical, and pesticide industries. In addition it is used to manufacture batteries, tetraethyllead, galvanized metal, lead shot, bullets, linotype metal, rubber, and plastics. Lead is not an essential element in the nutrition of human beings or animals. Although trace levels of lead are natural components of animal and plant life, lead must be considered as an extremely toxic substance (Browning, 1969). The USPHS maximum permissible level for the element in drinking water is 0.05 ppm although its total absence is recommended. In irrigational water the allowable tolerances for lead are 5 and 20 ppm for long and short term use respectively (Anon., 1968).

Lead occurs naturally throughout the United States, and its concentrations in surface and ground waters used for domestic supplies range from the limit of detectability to 0.04 ppm. The average is about 0.01 ppm (Anon., 1953). However, the concentrations of lead are higher in waterways near heavily traveled highways because of lead emissions from internal combustion engines. Additionally, substantial amounts of lead are being introduced into natural waters from motor boats. English, *et al.* (1963) reported that approximately 0.53 grams of lead are introduced into the water for each gallon of leaded gasoline burned. With these data Ettinger (1967) has estimated that approximately 880,000 pounds of lead were released into watercourses by outboard exhaust in the United States in 1963.

In the aquatic ecosystem lead salts can be toxic to fish in soft waters at concentrations of between 0.1 and 0.4 ppm as lead (Jones, 1938; Carpenter, 1925; 1927). In hard waters some fish appear to tolerate as much as 1 ppm of lead indefinitely



(Jones, 1938). In a recent study Brown (1968) reported the 96-hour  $LC_{50}$  for rainbow trout to be 1.0 ppm as lead. Additional data summarized in Water Quality Criteria (McKee and Wolf, 1963) show that concentrations of lead between 0.1 and 1.0 ppm are very toxic to a number of fish species.

#### Manganese:

Manganese is used to manufacture alloys, dry cell batteries, chemicals (as a catalyst), glassware, ceramics, paints, varnish, inks, dyes, matches, and fireworks, as well as in agriculture to enrich manganese deficient soils. Manganese is essential for the nutrition of both animals and plants; and its salts are toxic only at very high dosages (Browning, 1969). The USPHS drinking water standard of 0.05 ppm was established for esthetic and economic considerations rather than for physiological hazards. Since high levels of manganese in the soil can be phytotoxic, its irrigational tolerance limits have been set at 2.0 and 20.0 ppm for long and short term use respectively (Anon., 1968).

Manganese divalent and trivalent chloride, nitrate, and sulfates are water soluble and occur naturally in many soils. In solution these manganese ions form insoluble carbonates and hydroxides which limit manganese concentrations in natural waters to much less than 1.0 ppm. The reported concentrations of this element in typical surface and ground waters in the United States usually range between the limit of detection and 0.14 ppm (Durfer and Becker, 1964).

The toxicity of manganous and manganic ions to aquatic organisms is slight. Levels in the range of 1 ppm are tolerated by fish with few adverse effects. In many cases small amounts of manganese in the range of 0.0005 to 0.1 ppm have been reported to stimulate growth of certain aquatic organisms. However, Guseva (1937; 1939) found that concentrations of manganese in excess of 5 ppb exhibited a toxic effect on some algae. Permanganate ion is much more toxic to aquatic organisms than either manganic or manganous ions, but it is not stable in water and quickly decomposes.

### Mercury:

Mercury is or has been used in the manufacture of electrical apparatus, chlorine, caustic soda, industrial control instruments, paints, pharmaceuticals, cosmetics, paper, pulp, agricultural fungicides, and dental amalgams as well as in many minor industrial and consumer products. Moreover, the amalgamative and catalytic properties of mercury and its compounds are responsible for many additional industrial uses. Over three thousand (Bailey and Smith, 1964) distinct applications of mercury and its compounds have been found within these categories. Also, large quantities of mercury are released into the atmosphere from burning fossil fuels and smelting metal ore such as lead, zinc, and copper. All of these are associated with minute quantities of mercury.

Mercury is not an essential element for the nutrition of humans or animals; and its physiological role, if any, is unknown (Browning, 1969). Because mercury is a cumulative and extremely toxic poison in animals, it is of primary concern in the long term contamination of aquatic ecosystems. The USPHS maximum permissible level for mercury in drinking water is 0.005 ppm with a strong recommendation that none be allowed (Anon., 1968).

Mercury is a naturally occurring element that is normally found in minerals, rocks, soil, water, air, plants, and animals. The ubiquitous nature of mercury is due, in a large part, to the high vapor pressure of the element and its compounds. The background level of mercury ranges between 0.02 to 0.7 ppb in natural surface and ground waters away from known mercury deposits in the United States (Hinkle and Learned, 1969; Chau and Saitoh, 1970).

Until the middle 1960's mercury compounds and especially elemental mercury were presumed to simply be assimilated and diluted when they were released into the environment. In 1967, moderately toxic inorganic mercury was proven to be biologically converted into highly toxic methylmercury by sediment microbes. Moreover, methylmercury compounds can be biologically magnified

in aquatic organisms by factors of thousands over the mercury levels in the water. Therefore, the capability of aquatic microbes to methylate an inorganic ion (mercuric ion) and form methylmercury combined with the ability of aquatic organisms to concentrate methylmercury thousands of times has created the most serious heavy metal pollution problem known to date. Because mercury's unique contribution to the pollution problem has only recently been recognized the following review of its scientific literature is warranted.

*biological methylation --*

From an ecological standpoint, a most serious aspect of the environmental mercury problem is that selected microbes in an aquatic environment are capable of biologically synthesizing methylmercury from mercuric (II) ion (Jensen and Jernelov, 1969; Wood, *et al.*, 1968; Fagerstrom and Jernelov, 1971; Dunlop, 1971). The amounts of mono- and methylmercury compounds are a result of the population of microbial species, organic pollution loading, mercury concentration, temperature, and the pH of the system. At low mercury contamination levels dimethylmercury is the ultimate product of the methyl transfer reaction whereas if higher concentrations of mercury are introduced, monomethylmercury is produced (Wood, *et al.*, 1968). With regard to the pH, the data show that neutral and alkaline environments favor the formation of dimethylmercury (Wood, *et al.*, 1968; Larsson, 1970) which readily decomposes to the water soluble monomethylmercury form in mildly acidic environments. Therefore, the pH can be correlated with the mercury levels in a contaminated body of water.

*biological magnification --*

Not only are the methylated forms of mercury more highly toxic than other forms of mercury, but they are also more biologically mobile. Aquatic organisms are able to concentrate methylmercury either directly from the water or through the food chain (Johnels, *et al.*, 1967; Hannerz, 1968; Hasselrot, 1968; Miettinen, *et al.*,

1970). Thus, two biological processes are involved: the biological conversion of relatively low toxicity inorganic mercury compounds into highly toxic methylmercury and the biological magnification of this material in the aquatic organism.

While mercury concentrations greater than 10-20 ppb can be toxic to aquatic organisms, sublethal levels are also absorbed and biologically magnified. Mercury is accumulated after it is ingested or absorbed directly from the water through the organisms' outer surfaces (skin or epithelium) and/or across gill membranes during respiration. Since mercurials are at least one thousand times more soluble in lipids than in water, they are easily extracted from water or food by contact with the lipid portions of the tissues (Hughes, 1951). That 85-95% of the total mercury in contaminated fish is in the form of methylmercury is accounted for by its affinity for sulfhydryl groups and lipids (Westoo, 1966; Noren and Westoo, 1967).

Moreover, the magnitude of biological accumulation of mercury by aquatic organisms is a function not only of the species and its exposure interval but also of the feeding habits (trophic level), metabolic rate, age or size of the organism, and the various water quality parameters as well as the degree of mercury pollution.

Furthermore, the amount of inorganic and organic dissolved and particulate matter in an aquatic environment is significant in the accumulation of mercury by fish. D'Itri, et al. (1971) established that rainbow trout (Salmo gairdnerii) taken from an oligotrophic lake contained mercury levels three times greater than those of rainbow trout taken from a nearby eutrophic lake. Hannerz (1968) reported that if all other factors were equal, more mercury would accumulate in aquatic organisms as a function of their feeding habits and metabolic rates than their trophic level in the food chain.

Jernelov (1970) established that the amount of methylmercury released from the sediments depends not only on the depth at which it is buried but, more importantly, on the activity of the



macrofauna that live in the sediments. Without macroorganisms the formation and release of methylmercury occurs almost entirely in the top centimeter of the sediment. However, if aggressive bottom feeding organisms such as Anodonata are present, as much as nine centimeters of the surface layer of sediment is disturbed; and methylmercury is released into the water.

Under long term chronic exposure, if mercury is eliminated from an organism faster than it is accumulated, an equilibrium or steady state of mercury contamination is reached. In this state mercury levels are greater in the muscle tissue of the fish than in their liver or kidney (Wood, et al., 1968; Johnels, et al., 1967). If the mercury loadings increase, the organisms may build up mercury faster than they can eliminate it. This biological accumulation has been characterized by greater mercury levels in the liver and kidney tissues than in the muscles. Chapman, et al., (1968) reported that freshwater phytoplankton, macrophytes, and fish have the potential biological capability to magnify the mercury concentrations from water 1000 times while freshwater invertebrates have a 100,000 magnification factor. Johnels, et al. (1967) reported comparable biological magnification results of 3000 for northern pike.

Rucker and Amend (1969) established that rainbow trout exposed to 60 ppb of ethylmercury for one hour a day over ten days contained mercury levels of 4,000 and 17,300 respectively in their muscle and kidney tissues. Miettinen, et al. (1970) have shown that northern pike and rainbow trout are able to assimilate and concentrate into their muscle tissues methylmercury bound to ingested food objects. Also, several theories have been advanced to explain the mechanism by which methylmercury is concentrated at each trophic level of the food chain. One such theory is that a series of transfers take place during which the methylmercury from the water is taken up by the phytoplankton, ingested by zooplankton, and then consumed by forage fish. Hamilton (1971) has recently shown that the levels of mercury in fish food organisms increase at each trophic level of the food chain. Another theory



suggests that methylmercury and the bacteria that produce it are consumed by benthic organisms which are then preyed upon by bottom feeding forage fish which, in turn, fall prey to piscivorous fish. While the mechanism of mercury accumulation is not clear, it obviously is a function of one or more of the following: the metabolic rate in individual fish, differences in the selection of food objects as a fish matures, or the fish's epithelial surface area (Hannerz, 1968; Wobeser, *et al.*, 1970). In all likelihood the actual mechanism is a complex combination of these parameters.

#### *toxicity --*

Various reports have summarized the toxicity of mercury compounds to aquatic organisms (McKee and Wolf, 1963; Ellis, 1937; Doudoroff and Katz, 1953; Chapman, *et al.*, 1968; Anon., 1971; Nelson, 1971; D'Itri, 1972; Wallace, *et al.*, 1971; Boetius, 1960; Anon., 1970). The acute toxic action of mercuric ions results from damage to the gill tissues and/or the formation of a film of coagulated mucus that fills the interlamellar spaces and prevents the normal movement of the gill filaments. Therefore, the necessary contact between the gill tissues and the water is interrupted, and the gaseous exchange is impeded to such an extent that the fish die from asphyxiation (Carpenter, 1925). Lloyd (1960) attributes this asphyxiation to a breakdown and swelling of the gill epithelium that obstructed the ability of the gills to exchange gases. The chronic and subchronic toxic action of mercury compounds, especially lipid soluble organomercurials, may not react destructively with the gill membranes because of the extremely low levels of mercury.

Therefore, more of the mercurial is absorbed into the blood and other internal tissues of the organism. Moreover, if the various forms of mercury react in aquatic organisms as in mammals and birds, similar distribution patterns and elimination rates for inorganic and organic mercurials could be expected. Because of their higher solubility in lipids, however, the alkylmercurials,

especially methylmercury, are accumulated in the red blood cells and the central nervous system. Also, since alkylmercurials are inherently stable, the degradation of methylmercury to the mercuric form required for elimination from the aquatic organisms is minimal.

Although literature is abundant on the toxicity of mercury in the aquatic environment, these reports often differ widely. Many of the data relate to acute toxicity experiments of extremely short duration. Furthermore, the wide range of concentrations at which mercury is toxic to aquatic organisms indicates that the chemical and physical factors as well as the normal environmental variables affect the degree of toxicity of a particular mercurial. In general, mercury toxicities have been shown to be a function of several water quality parameters: temperature, pH, organic pollution loadings, hardness, alkalinity, heavy metal loadings, and dissolved oxygen. Furthermore, the relative toxicity of mercury to aquatic organisms varies widely with the species, its life stage, and the state of adaptation of the animals' acclimation to the environmental conditions in the aquatic ecosystem. Therefore, any estimation of the toxicity of mercury or any of its compounds to aquatic organisms is extremely difficult. Carpenter (1927) stated that there is no theoretical lower limit for the toxicity of the salts of heavy metals, and even trace amounts of mercuric chloride would be toxic to fish if the exposure were long enough. Boetius (1960) supported this conclusion and established that the product of the concentration of mercuric ion and the survival time of a given species is a constant. The constant is a function of the species, body weight, life stage, and the physical and chemical properties of its aquatic ecosystem.

Boetius also contended that the lethal quantity of mercury is in direct proportion to the body weight of the fish, presumably as a function of the surface area of the body and gills. Therefore, the organism must accumulate a certain quantity of the mercurial to be lethal.

With respect to the toxicity of inorganic mercury in the form of mercuric ion, short term studies indicate that concentrations in

the range of 1 ppm are fatal to fish (Jones, 1939; Boetius, 1960; Weir and Hine, 1970). For long term exposures of 10 days or more, mercury levels as low as 10 to 20 ppb have been shown to be fatal to fish. Uspenskaya (1946) established that concentrations of 10 and 20 ppb of mercury were fatal to *Phoxinus* in 80-92 and 19-32 days respectively. Belding (1927) established that 12.5 ppm was the lowest concentration of mercuric chloride that was fatal to brook trout within 24 hours. However, Rushton (1920) reported that 10 ppm of mercuric chloride killed rainbow trout (*Salmo gairdnerii*) in only 15 minutes.

Since 1914 when organomercurials were first introduced into agriculture, increasingly larger quantities of them have been lost to the environment. While these organomercurials are generally more toxic to aquatic organisms than inorganic mercury compounds are, fish are able to survive relatively higher concentrations with little ill effect for short periods of time. For example, Rucker and Whipple (1951) report that steelhead trout (*Salmo gairdnerii*) fry and three inch blueback salmon (*Oncorhynchus nerka*) are able to survive in 10 ppm of pyridylmercuric acetate with no toxic effects for one hour. Willford (1966) reported on 24 and 48 hour  $LC_{50}$  values for pyridylmercuric acetate concentrations that range between 2.6 to 16 ppm and 2.5 to 12.8 ppm respectively for six different species of freshwater fish. Clemmens and Sneed (1958; 1959) published data on an extensive series of bioassay studies conducted to determine the effect of pyridylmercuric acetate on channel catfish fingerlings (*Ictalurus punctatus*). At  $LC_{50}$  values of 0.39 the fish lasted 72 hours whereas at 3.2 ppm they endured only 24 hours.

Hamamoto (1960a) compared the tolerance of carp (*Cyprinus carpio*) to phenylmercuric acetate, propionate, butyrate, and valerate. He reported  $LD_{50}$  values that ranged from 154 to 275 ppm for 24 hours and 76 to 208 ppm for 48 hours. The increased toxicity of alkylmercurials over arylmercurials was established in another study by Hamamoto (1960b). The 24 and 48 hour  $LD_{50}$  values for the homologous series of ethyl, propyl, butyl, and amylmercuric chloride

were found to range between 43-132 ppm and 8-75 ppb respectively.

On the basis of 120 hour bioassay tests conducted on minnows (Notropis cornutus, Notropis whipplii, and Notropis atherinoides), Van Horn and Balch (1955) determined that the minimum lethal concentrations of ethylmercuric phosphate, pyridylmercuric acetate, pyridylmercuric chloride, and phenylmercuric acetate are 800, 150, 40 and 20 ppb respectively. Rucker and Amend (1969) established that ethylmercuric phosphate has a greater rate of gill absorption than pyridylmercuric acetate, and each mercurial forms a different distribution in the various tissues of the fish. In a later paper Amend, et al. (1969) explained that these biological absorption differences are primarily due to the different chemical properties as well as to the physiological degradation products of each mercurial. Moreover, the poisonous effects on rainbow trout (Salmo gairdnerii) of a single large exposure to ethylmercuric phosphate (125 ppm for one hour) increased with an increase in the water temperature or the chloride ion concentration or with a decrease in the dissolved oxygen content of the water (Amend, et al., 1969).

The distribution of methylmercury in pike (Esox lucius) was investigated by Backstrom (1969) and Miettinen, et al. (1970). Miettinen, et al. (1970) estimated the LD<sub>50</sub> value of methylmercury for northern pike to be  $15 \pm 3$  mg per kilogram of the fish's live body weight when the methylmercury was administered as a single dose in food. Furthermore, when administered in a single dose, 12 ppm is toxic to rainbow trout in 1.5 to 2 hours. However, when the same amount was fed to the fish as two portions with an interval of from one to two days, only 25 per cent of the fish died after 94 days. These experiments also established that the cause of death for these fish was kidney failure. Miettinen (1970) also reported that the chemical form of methylmercury administered to the fish made little difference in its toxic effect. Moreover, when the methylmercury is bound to sulfhydryl groups of proteins as it would be received in nature, it is just as toxic as the free unbound ionic form. Therefore, through gill absorption and long term ingestion of contaminated food, fish accumulate much higher



levels of methylmercury than they could tolerate in a single large dose.

While aquatic plant tissues appear to have no appreciable uptake of mercury, Hannerz (1968) established that the mercury levels were significantly higher in the submerged portions because of surface adsorption. However, Johnels, et al. (1969) reported levels as high as 140 ppb in the freshwater algae (Nitella). Therefore, while in effect, no mercury is translocated from the water into the plants, animals are able to filter and accumulate any mercury that is present in the aquatic ecosystem.

Johnels, et al. (1969) examined various invertebrates to determine their accumulation of mercury. The midge (Chironomus plumosus) accumulated 230 ppb; the stonefly (Isoperla) had 72 ppb; 65 ppb in the sowbug (Asellus aquaticus), 49 ppb in the burrowing alderfly (Sialis); 140 ppb in zooplankton and 440 ppb in the tail muscle of a crayfish. In an area receiving mercury pollution these investigators found that the burrowing alderfly, sowbug, and caddisfly had mercury levels of 5500, 2400, and 1700 ppb respectively. Water fleas (Daphnia magna) are more susceptible to mercuric chloride than fish. For example, six ppb could immobilize the Daphnia in less than 64 hours (Anderson, 1948). Bringmann and Kuhn (1959) reported 30 ppb as the threshold toxicity level of mercuric chloride for Daphnia. In other studies of invertebrates Hannerz (1968) reported that freshwater snails (Planorbis sp. and Lymnaea stagnalis) accumulate higher levels of phenyl, methoxyethyl, and methylmercury than of mercuric chloride. And Jones (1950) found that the flatworm (Polycelis nigra) has a 48 hour toxicity threshold of 200 ppb for mercuric chloride as mercury.

Getsova and Volkova (1964) investigated the ability of various insects to accumulate radioactive mercury-203. They found biological magnification ratios relative to the water concentration of mercury to be 4000 for the dragonfly (Aeschna grandis), of 5240 for the midge (Glyptotendipes punctatolineatus), and of 216 for the fly (Eristalis tenax). In a later study Warnick and Bell (1969)



established 96 hour  $TL_m$  values of 2.0 ppm as mercuric ion for the stonefly (Acroneuria lycorias), mayfly (Ephemerella subvaria), and the caddisfly (Hydropsyche betteni). These insects appear to be much less sensitive to the toxic effects of mercury than many fish are.

Harriss, et al. (1970) established that concentrations as low as 0.1 ppb of selected organomercury fungicides decreased both the photosynthesis and the growth of laboratory cultures of marine Nitzschia delicatissum as well as of some freshwater phytoplankton species. Ukeles (1962) reported that ethylmercuric phosphate was lethal to marine phytoplankton at 60 ppb, and levels of less than 0.6 ppb drastically limited the growth of these organisms.

#### Molybdenum:

Molybdenum is used to manufacture alloys, glassware, ceramics, printing ink, lacquers, and paints as well as electrical and electronic components. It is also used as a catalyst in the chemical industry and in agricultural fertilizers for leguminous crops (Browning, 1969).

Molybdenum has been recognized as an essential micronutrient for plants since 1939 (Arnon and Stout, 1939). It is essential to the nutrition of all nitrogen-fixing organisms. Molybdenum exhibits a low degree of toxicity in some animals; thus it is only considered toxic in rare instances (Browning, 1969).

The concentration of molybdenum in natural waters is often below 1 ppb although levels as high as 100 ppb (Durum and Haffty, 1961) have been reported for waters in molybdenum rich drainages. The USPHS has not set a mandatory permissible limit for this element in public water supplies or for farmstead water supplies (Anon., 1968). Tolerance limits of 0.005 and 0.05 ppm for long and short term irrigation waters have been recommended because legumes can concentrate molybdenum in their tissues to levels in the order of 5 ppm (Kubota, et al., 1963). Molybdenum levels in this range are toxic to cattle.

In natural waters molybdenum is probably not a hazardous element since 96-hour  $LM_{50}$  values for the fathead minnow were only 70 and 370 ppm in soft and hard water respectively (Tarzwell and Henderson, 1956). *Daphnia* have been reported to withstand up to 1000 ppm without ill effects (Bringmann and Kuhn, 1959). With respect to biological concentration, Lackey (1959) reported that several species of algae can concentrate molybdenum from the water by a factor of 2 to 15 times.

#### Nickel:

Nickel is used in the pesticides, electroplating, ceramics and glassware industries. It also acts as a catalyst in selected chemical processes as well as in the manufacture of various metal alloys and nickel-cadmium batteries (Browning, 1969). Probably burning fossil fuels contributes most of the nickel to the atmosphere (Sullivan, 1969). In addition, however, nickel carbonyl additives in gasoline are a source of air contamination.

Nickel and its soluble salts are toxic to animals in large doses and therefore do not constitute a health hazard at the levels encountered from environmental pollution sources.

The nickel concentrations reported in large rivers of North America ranged between 0.0 and 71 ppb with a median value of 10 ppb (Durum, 1971). Additionally, the USPHS has not recommended a mandatory permissible limit for this element in public water supplies or for farmstead water supplies (Anon., 1968). Tolerance limits of 0.5 and 2.0 ppm have been recommended for long and short term irrigation waters since the growth of some crops such as citrus plants, flax, sugar beets, tomatoes, potatoes and oats have been inhibited (Vanselow, 1952; Hewith, 1953).

Nickel has been reported to be less toxic to fish than zinc, copper, or iron (Podubsky and Stredronsky, 1948). However, other authors found nickel to be more toxic than either iron or manganese (Kalmmerer and Erickson, 1951). Recent studies by Pickering and Henderson (1964) established that the 96-hour  $LC_{50}$

of nickel for fathead minnows ranges from 5 ppm in soft water to 43 ppm in very hard waters. These data corroborate Jones' (1939) findings that fish were able to survive in waters receiving mining waste drainage which contained nickel concentrations between 13 and 18 ppm. Thus, depending on water quality parameters such as hardness as well as synergistic and antagonistic factors, the safe levels of nickel for fish and other aquatic organisms in natural waters is estimated to be 0.1 to 1.0 ppm (Doudoroff and Katz, 1953).

#### Selenium:

Selenium is used industrially as a catalyst for chemical processes and in the production of electronic components, such as semiconductors, photoelectric cells and rectifiers. It is also used to manufacture paints, dyes, glass, pesticides, plastics and rubber (Browning, 1969). Significant amounts of selenium are also released into the atmosphere as water soluble selenium dioxide from the burning of fossil fuels.

Selenium is an essential element for animal and plant nutrition. However, excess amounts can be toxic to animals and plants. The USPHS drinking water permissible tolerance level is 0.01 ppm with the recommendation that it be absent (Anon., 1968). The recommended levels in waters used for farmstead and livestock purposes has been set at 0.01 ppm or less. The maximum allowable levels of selenium in water used for either long or short term irrigational purposes has been set at 0.05 ppm (Anon., 1968). This level was instituted because some plants accumulate selenium levels of 4-5 ppm from waters containing 0.05 ppm of dissolved selenium. These higher levels in plants can cause toxic effects in animals that eat them.

Selenite ( $\text{SeO}_3^{=}$ ) and Selenate ( $\text{SeO}_4^{=}$ ) ions are the chemical species most likely to be found in natural waters. However, very few data are available on the background levels in natural waters. It is estimated that in natural waters receiving no industrial pollution the levels of selenium are less than 1 ppb. The toxic

effects of selenium in the aquatic ecosystem are not very well known. For example, levels of 2.0 ppm of selenium as sodium selenite have been reported to exhibit toxic effects on goldfish in 8 days and lethal effects in 18 to 46 days (Anon., 1950). In another study Bringmann and Kuhn (1959) established the 48-hour  $TL_m$  for Daphnia to be 2.5 ppm. Copeland (1970) has also reported that some species of zooplankton have the ability to concentrate selenium biologically at least up to levels of 7 ppm.

#### Silver:

Silver is used in the manufacture of silverware, jewelry, alloys, brazing alloys, solders, antiseptics and photographic film. Silver is also used in electroplating, inks, coloring porcelain, and in the processing of food and beverages (Browning, 1969).

Silver is not considered an essential element for either plant or animal nutrition and is not normally found in animal or human tissues (Browning, 1969). Silver and its salts are moderately toxic, and the USPHS maximum permissible concentration of silver in drinking water supplies was set at 0.05 ppm for cosmetic rather than public health considerations (Anon., 1968). The recommended limit of silver in water for general farmstead uses is also 0.05 ppm. However, no tolerance limits have been established for this element in irrigation waters (Anon., 1968).

The presence of silver in natural waters containing chloride ions is minimized because silver chloride is insoluble, and levels in excess of 10 ppb are rare. Silver is toxic to aquatic life, and lethal concentrations in the range of 4 to 50 ppb have been reported (Murdock, 1953; Doudoroff and Katz, 1953; Anderson, 1948; Shaw and Lawrance, 1956). Also, Anderson (1948) found that silver concentrations as low as 4 ppb immobilized Daphnia.

#### Tin:

Tin is used in textile mordants, reducing agents in chemical processes and stabilizers in plastics and paints. It also has

application in finger nail polishes, alloys, fungicides, insecticides, poultry antihelminthics, glass works, and to decorate porcelain. In addition, electro-deposited tin coatings are used in electrical, radio, and engineering components, and in the pistons of internal combustion engines (Browning, 1969).

Tin exhibits a low order of toxicity towards animals, and no data suggest that tin is an essential element for animal or plant nutrition. No maximum permissible levels in drinking waters have been established by the USPHS nor have recommended levels been suggested for farmstead, livestock, or irrigation waters (Anon., 1968).

Tin is rarely found in natural waters because many of its salts are extremely insoluble. Urban and industrial pollution sources are responsible for most of the trace levels of tin in surface and ground waters. Few data are available concerning toxic levels to aquatic organisms. Goldfish have been reported to have survived in water containing 1000 ppm of stannous chloride for 1 to 1.5 hours in soft water and up to 5 hours in hard water (Doudoroff and Katz, 1953; Anderson, 1948). Levels 146 ppm of stannic chloride will immobilize Daphnia (Anderson, et al., 1948) whereas the toxicity threshold of stannous chloride toward Daphnia is only 25 ppm (Anderson, 1948). Finkel and Allee (1940) reported that about 0.6 ppm stannous ions were beneficial to goldfish in that their growth was accelerated. Therefore, it appears that the stannous form of tin is more toxic to aquatic organisms, and stannous levels of less than 1.0 ppm may be tolerated with few or no deleterious effects.

#### Titanium:

Titanium is used in the manufacture of metal alloys, paint pigments, electronic equipment, glassware and ceramics. It is also used as a filter in paper making and as a mordant in the textile industry (Browning, 1969).

No evidence suggests that titanium is an essential nutrient for plant and animal nutrition. However, it does exhibit a very low toxicity toward animals. Maximum permissible levels in drinking



waters have not been established by the USPHS nor have recommended levels been suggested for farmstead, livestock, or irrigation uses (Anon., 1968).

In natural waters, levels of titanium are usually no more than 1 ppb because titanium compounds in the aquatic environment have limited solubility. Thus, titanium does not appear to be a serious toxicant. The 96-hour  $TL_m$  for fathead minnows in soft and hard water was established to be 8.2 and 120 ppm respectively (Tarzwell and Henderson, 1956). However, titanium may be biologically concentrated in aquatic organisms from the water. Concentration factors of between 200 and 10,000 have been reported for six species of brown algae (Lackey, 1959).

#### Vanadium:

Vanadium is used to manufacture glassware, dyes, inks, paints, varnish dryers, alloys, and pesticides. It is also used as a catalyst in chemical processes, as a mordant in dyeing, in printing fabrics, and in the photographic industry (Browning, 1969). Vanadium associated with petroleum can be emitted into the atmosphere when diesel and fuel oils are burned (Athanassiadis, 1969). Vanadium is also emitted from such sources as industries producing the metal, its chemical compounds, alloys, and other products as well as power plants and utilities that consume residual and crude oils and coal containing vanadium.

Vanadium is moderately toxic to animals as it has a low order of oral toxicity, but it does not affect the growth or life span of test animals. While vanadium appears to be an essential element required for the metabolism of some higher green plants, it has not been proven necessary for animal metabolism even though some data indicate that it may be required for certain biological functions (Browning, 1969). Levels of vanadium in drinking waters at concentrations of 0.03 to 0.22 ppm have been reported to have no noticeable effect on human beings and, in fact, was shown to reduce the incidence of dental caries (Tank and Storvick, 1960).

While vanadium is not normally present in surface waters (Durum, 1971) levels as high as 0.15 ppm have been reported in waters of selected areas of New Mexico (Stokinger, 1960). The USPHS has not set a mandatory permissible limit for this element in public water supplies nor have recommended levels been established for farmstead or livestock water supplies (Anon., 1968). The recommended tolerance for vanadium in irrigational waters for both long and short term usage has been set at 10 ppm. This limit was set because vanadium has induced toxic effects in several plant species that received water containing between 10 and 20 ppm of the element (Warington, 1951).

In the aquatic ecosystem vanadium does not appear to be excessively toxic. Tarzwell and Henderson (1956) reported 96-hour  $TL_m$  values for vanadyl sulfate (as vanadium) to be 4.8 and 30 ppm in soft and hard waters respectively. Vanadium pentoxide was found to be less toxic with 13 and 55 ppm needed to produce the same effects. For bluegills the 96-hour  $TL_m$  values for vanadyl sulfate (as vanadium) were 6 and 55 ppm in soft and hard waters respectively.

#### Zinc:

Zinc is used in the manufacture of galvanized products, dry cell batteries, alloys such as brass, dyes, paints, pesticides, cosmetics, pharmaceuticals, and many other products. Zinc is an essential element for animal and plant nutrition, and it exhibits a relatively low degree of toxicity to animals (Browning, 1969).

The USPHS maximum permissible limit for this element has been set at 5.0 ppm with the recommendation that it be virtually absent (Anon., 1968). The recommendation for zinc concentrations in farmstead waters is also 5.0 ppm with no levels established in waters for livestock use. The tolerance for zinc in irrigation waters has been set at 5 and 10 ppm for long and short term uses. These levels were set because zinc has produced toxic effects in selected plants when the concentrations ranged from 3 to 10 ppm

(Chapman, 1966).

In surface waters zinc may be present in trace amounts which are usually less than 5 to 10 ppb. A recent study of large rivers in the United States showed that the zinc concentrations ranged between 0.0 and 215 ppm with a median value of less than the detectable limits of 10 ppb.

While data show that human beings and animals have used water containing up to 50 ppm zinc with no ill effects (Hinman, 1938; Anderson, *et al.*, 1934; Barton and Weigle, 1932), aquatic organisms have been severely affected at much lower concentrations. Pickering and Henderson (1964) reported the 96-hour  $LC_{50}$  of zinc for fathead minnows at 0.87 and 33 ppm respectively in soft and hard water. Bluegills reportedly were able to withstand higher levels of zinc than the fathead minnows. Brungs (1969) established that zinc levels of 0.18 ppm in hard water were sufficient to cause an 83% reduction in the eggs of the fathead minnow. The estimated "safe" concentration for these fish was 0.03 ppm. Affleck (1952) reported that 0.04 ppm zinc in soft water prevented rainbow trout eggs from hatching. On the other hand, Goodman (1951) found that rainbow trout eggs hatched successfully in the presence of as much as 1 ppm zinc. The relative toxicity of zinc salts is controlled to a large degree by the hardness of the water. In soft waters, lethal levels of zinc salts in fish are between 0.1 and 1.0 ppm. The presence of calcium salts in the water has a great influence on the toxicity of zinc. Jones (1938) found that the lethal concentration for zinc to mature fish in water which contained only 1 ppm calcium was 0.3 ppm. However, when the calcium concentration was increased to 50 ppm, the fish could tolerate as much as 2.0 ppm zinc with no ill effects.

#### Southeastern Michigan:

Heavy metals were found to enter the aquatic environment of Southeastern Michigan from the following sources:

1. Through natural weathering conditions soluble species leach out of geological formations materials. Theoretically,

heavy metals from this source should constitute their natural background concentrations in Southeastern Michigan. While data on these heavy metal concentrations are scarce, it appears that only aluminum, iron, zinc and possibly barium, have natural levels greater than 100 ppb (See Tables 1 and 2 in Appendix-A). All other natural heavy metals have smaller concentrations.

2. Heavy metals discharged directly into receiving waters from industrial operations.

The estimated average daily and yearly release of heavy metals from industrial sources into Southeastern Michigan is summarized in Table 23. As required by the Refuse Act of 1899, these data were disclosed by each industry that discharges wastes into navigable waters in Southeastern Michigan. A tabulation of these heavy metal discharges by company and watershed is presented as Table 3 in Appendix-A. The information is not complete because some companies elected to classify these data as confidential information which was not available without special permission.

Attempts to estimate the total yearly contribution of heavy metals to waterways by individual industries were also hampered by three other problems. First diurnal variations are associated with batch production. Second, not only do daily variation occur, but there is little flow on holidays and weekends. Third, seasonal variations depend on what raw materials are available.

3. Discharge of heavy metals into receiving water from sewage treatment plants.

Table 24 summarizes the estimated average yearly release of heavy metals associated with wastewater discharges into Southeastern Michigan surface waters. The heavy metal concentrations were estimated on the basis of the heavy metal analyses of selected municipal treatment plant influents and effluents (Table 4, Appendix-A).

While large quantities of heavy metals pass through the sewage treatment plants and then into the surface waters of the Southeastern Michigan area, great amounts of these metals are also adsorbed and removed with the activated sludge. A pilot

Table 23. Estimated average daily and yearly release (in pounds) of heavy metals into Southeastern Michigan surface waters from industrial sources.

Heavy Metal	Daily Release	Yearly Release*
Aluminum	4,551	1,183,172
Antimony	896	232,960
Arsenic	31	7,953
Barium	3,384	879,840
Cadmium	21	5,582
Chromium	233	60,580
Cobalt	2.0	520
Copper	2,086	542,360
Iron	30,220	7,857,200
Lead	689	179,140
Manganese	1,660	431,600
Mercury	10	2,600
Molybdenum	3	780
Nickel	217	56,420
Tin	44	11,403
Titanium	266	69,160
Zinc	295	768,820
	44,608	12,290,090

\*Based on 260 working days per year.



Table 24. Estimated average yearly release of heavy metals into Southeast Michigan's surface waters associated with wastewater discharges. The total volume of municipal wastewater treatment plants being estimated at 1000 MGD.

Heavy Metal	Assumed Heavy Metal Concentration (ppb)	Yearly Release (Pounds)
Aluminum	1000	1,967,706
Antimony	ID	--
Arsenic	10	19,677
Barium	20	39,354
Cadmium	19	37,386
Chromium	360	--
Cobalt	ID	--
Copper	190	373,864
Iron	9800	19,283,520
Lead	100	196,771
Manganese	50	98,385
Mercury	3.0	5,904
Nickel	410	806,760
Selenium	10	19,677
Silver	ID	--
Tin	10	19,677
Titanium	ID	--
Zinc	740	<u>1,456,103</u>
TOTAL		25,033,158

ID = Insufficient Data

study of a Detroit sewage treatment plant determined the concentrations of selected heavy metals in the activated sludge (See Table 25). These data indicate that selected heavy metals can be concentrated 10 to 16 thousand times the levels in the final effluents depending on the composition of the activated sludge. Therefore, where land application is recommended for sludge disposal, the reactions and translocation of these heavy metals must be carefully considered.

4. Atmospheric fallout of heavy metals resulting from man's industrial, urban, and recreational activities.

Potentially large quantities of heavy metals may be released into the atmosphere of industrial cities. For example, in the Chicago area the amount of cadmium, copper, and lead released into the air has been estimated to be 12, 133, and 2152 tons per year respectively (See Table 5, Appendix-A). Estimates of the contributions of various airborne heavy metals to the Southeastern Michigan atmosphere are presented in Table 26. These airborne heavy metal concentrations are estimated on the basis of data presented in Tables 5 - 8, Appendix-A. To arrive at the approximation that 6,000,000 pounds of heavy metals have been transferred to the land surface each year, a complete turnover of the airborne heavy metals was assumed to have occurred every 3.6 days. Nevertheless, since the estimated contribution of airborne heavy metals is about one-half of the yearly release from industrial sources (See Table 23), it must be considered a significant source of heavy metals to the Southeastern Michigan area.

5. Miscellaneous discharges.

This category includes heavy metals such as aluminum, iron, titanium, arsenic, cadmium, chromium, copper, manganese, nickel, lead, selenium, zinc, and tellurium that are leached from coal ash sluiced into the holding ponds. Very little information is available on this kind of water pollution (Rohrman, 1971). A related source of water pollution is land runoff of agricultural pesticide preparations which contain heavy metals, but few data are available to indicate the amount of heavy metals contributed to Southern Michigan from this source.

Table 25. Concentration of heavy metals in the activated sludge of a Detroit sewage treatment plant pilot study (Shannon, 1972).

Element	Concentration (ppm on dry weight basis)			
	January, 1972	June, 1972	July, 1972	Milorganite*
Chromium	730	5,730	1,945	6,500
Iron	33,000	135,000	65,200	49,000
Copper	415	2,080	666	620
Nickel	1,620	4,070	1,165	150
Cadmium	40	226	139	1,700
Zinc	136	8,330	2,600	1,650

\*Milorganite is a commercial fertilizer composed of dried raw and activated sludge.

Table 26. Estimated contribution of airborne heavy metals to the Southeastern Michigan area.  
(Assuming a 500 sq. mile industrial area 2000 meters in height, i.e. approximately  $2.6 \times 10^{12}$  cubic meters of atmosphere).

Heavy Metal	Assumed Heavy Metal Concentration (ng/m <sup>3</sup> )	Static Quantity of Heavy Metal in Atmosphere (Pounds)	Amount Transferred to land surface (Pounds/Year)
Cadmium	19	109	10,900
Lead	200	11,400	1,140,000
Copper	600	3,430	343,000
Iron	4800	27,400	2,740,000
Aluminum	2000	11,400	1,140,000
Zinc	367	2,100	210,000
Manganese	153	875	87,500
Titanium	184	1,050	105,000
Chromium	33	189	18,900
Antimony	9	51.4	5,140
Arsenic	5	28.6	2,860
Silver	1.0	5.7	570
Mercury	2.3	13.2	1,320
Selenium	2.3	13.2	1,320
Cobalt	1.2	6.9	690
Nickel	100	570	57,000
Molybdenum	20	114	11,400
Vanadium	200	1,140	11,400
Barium	600	3,430	343,000
Beryllium	1.0	5.7	570
Tin	20	114	11,400
			<hr/> 6,241,970

\* Assuming a complete turnover every 3.6 days.

The concentration of dissolved heavy metals measured in the waters of Southeastern Michigan have been tabulated (Tables 9 - 15, Appendix-A) and are summarized in Table 27. These data show that the average dissolved concentration of all the heavy metals measured, with the exception of zinc, were less than 10 ppb. For mercury and cadmium the average measured levels were normally less than 0.1 and 1.0 ppb respectively.

Table 27. Summary of heavy metal data for Southeastern Michigan Rivers.

Element	Average Value µg/l	Range µg/l
Arsenic	<10	<2-30
Cadmium	< 1	0-11
Chromium	< 1	0-3300
Cobalt	< 1	1
Lead	<10	2-25
Zinc	32	0-500
Mercury	< 0.1	<0.02-0.6
Copper	8	0-70
Nickel	7	0-50
Antimony	0	0

#### Hydrology and Materials Loading in Impact Zone

The Great Lakes hydrologic system represents a complexity which has had relatively long-term programs of surveillance by various agencies of those national territories within the watershed. This data base consists of precipitation, stream flow and lake level records dating from the previous century in certain cases. We undertook a lengthy data acquisition effort which resulted in the development of records of stream flow in the Great Lakes system bounded by the entrance to St. Clair River at Lake Huron, intermediate, tributary watersheds from southeastern Michigan, southwestern Ontario, Canada streams, and remaining United States watersheds terminating at the Lake Erie discharge to the

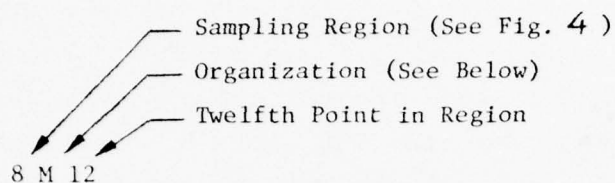


Niagara River. During this investigation we were able to identify nine agencies that had existing surveillance programs for stream flow and/or water quality. They include: Environment Canada; U.S. Geological Survey; Ontario Ministry of the Environment; Michigan Water Resources Commission; U.S. Environmental Protection Agency; and Pollution Control Agency, Toledo.

While hydrologic and water quality records were analyzed over a fairly long period, the data base for materials balance calculations were developed over the latest three-year period, 1969 through 1971. This decision was based on the following reasoning: (1) the volumetric residence time in the impact zone is approximately three years; (2) the data file prior to 1969 was not as extensive; and (3) this duration of analysis could be accomplished satisfactorily with the limits of our project progress schedule.

As was briefly mentioned in Section II of this report (pg. 8 - 10) we subdivided the impact zone into nine regions (Figure 4). Within these regions we were able to identify over 240 existing sampling stations. The exact location of each station and a station code number, based on sampling region and agency, is detailed in Table 28. When possible we identified the stations by degrees, minutes and seconds of latitude and longitude. This provided us with the flexibility to superimpose a particular station on a wide variety of maps. Some stations, however, were more accurately described by listing their upstream or downstream distance from a prominent landmark such as a bridge. When a sampling station consisted of two or more sampling points it is so indicated in the table.

STATION DESIGNATION FOR TABLE 28



<u>Organization</u>	<u>Measurement</u>	<u>Code</u>
Environment Canada (EC)	Streamflow	CA
	Water Levels*	CAN
United States Geological Survey (USGS)	Streamflow	US
	Water Quality	U
Ontario Ministry of the Environment (OME)	Water Quality	O
Michigan Water Resources Commission (MWRC)	Water Quality	M
Environmental Protection Agency (EPA)	Water Quality	E
Pollution Control Agency Toledo (TPCA)	Water Quality	T
Institute of Water Research (IWR)	Water Quality	I
Joint USGS-EPA	Water Quality	U-E
Detroit Metropolitan Water Department (DMWD)	Water Quality	D
National Ocean Survey-Lake Survey Center (NOAA)	Water Levels*	N

\*Streamflow in connecting waterways derived from water levels at selected stations in Gt. Lakes and connecting waterways.

Table 28. Descriptions of locations of streamflow gaging stations and water quality sampling points.

STATION*	ENVIRONMENT CANADA (EC)
4CAN118	Amherstburg-Station No. 11995: Ontario, lat. 42° 8.66', long. 83° 6.87'
4CAN119	Bar Point-Station No. 12005: Ontario, lat. 42° 3', long. 83° 6.66'
4CA11	Ruscom River near Ruscom Station
9CA14	Thames River at Thamesville, lat. 42° 32' 42", long. 81° 58' 04"
9CA17	Sydenham River near Dresden, lat. 42° 35' 38", long. 82° 06' 31"
9CA18	Sydenham River - Bear Creek above Wilkesport, lat. 42° 45' 52", long. 82° 20' 30"
9CA19	Sydenham River - Bear Creek near Petrolia, lat. 42° 54' 21", long. 82° 07' 12"
9CA22	Thames River - Dingman Creek at Lambeth, lat. 42° 54' 30", long. 81° 18' 20"
9CA23	Thames River at Byron, lat. 42° 57' 47", long. 81° 19' 55"
9CA24	Sydenham River at Alvinston, lat. 42° 49' 08", long. 81° 51' 30"
9CA25	Sydenham River at Strathroy, lat. 42° 57' 35", long. 81° 37' 42"
U.S. GEOLOGICAL SURVEY (USGS)	
2US9	Maumee River at Waterville, lat. 41° 30' 00", long. 83° 42' 46", on downstream side of second pier from left end of bridge on State Highway 64, 3 miles downstream from Tontogany Creek, and 21.1 miles upstream from mouth.
3U11	Lake Erie near Pointe Aux Peaux, lat. 41° 56' 20", long. 83° 14' 20", at Monroe Municipal Water Intake.

\*See designation on previous page.

- 4U24 Detroit River, Livingstone Channel below Amherstburg,  
lat. 42° 04' 03", long. 83° 07' 50"
- 4U25 Detroit River, Trenton Channel at Swan Island,  
lat. 42° 05' 43", long. 83° 10' 55"
- 8US4 River Raisin near Monroe, lat. 41° 57' 40", long.  
83° 31' 55", Monroe County, on left bank 0.8  
miles downstream from bridge on Ida Maybee Road,  
5.0 miles downstream from Saline River, and 7.5  
miles west of Monroe.
- 8US7 Huron River at Flat Rock, at highway bridge below  
Metler's Dam. (Continuous records 1904-1911 only).
- 8US9 Huron River at Ann Arbor, lat. 42° 17' 10", long.  
83° 44' 00", in NW 1/4 sec. 28, T.2S., R.6E.,  
Washtenaw County, on left bank 100 ft. upstream  
from bridge on Wall Street in Ann Arbor, 0.7  
mile downstream from Argo Dam, and 4.2 miles up-  
stream from Geddes Dam.
- 8US15 Lower River Rouge at Inkster, lat. 42° 18' 00",  
long. 83° 18' 00", in S 1/2 sec. 19, T.2S.,  
R.10E., Wayne County, on right bank 10 ft. down-  
stream from bridge on John Daly Road, 0.6 mile  
northeast of Inkster, and 4.8 miles upstream from  
mouth.
- 8US16 Middle River Rouge near Garden City, lat. 42°  
20' 55", long. 83° 18' 45", in W 1/2 sec. 6,  
T.2S., R.10E., Wayne County, on right bank 200  
ft. downstream from bridge on Inkster Road, 1.8  
miles northeast of Garden City, and 6.0 miles up-  
stream from mouth.
- 8US17 River Rouge at Detroit, lat. 42° 22' 20", long.  
83° 15' 20", in SW 1/4 sec. 27, T.1S., R.10E.,  
Wayne County, on right bank 500 ft. upstream from  
bridge on Plymouth Road in Detroit, and 4 miles  
upstream from Middle River Rouge.
- 8US19 Clinton River at Mount Clemens, lat. 42° 35' 45",  
long. 82° 54' 35", Macomb County, on left bank 20  
ft. downstream from bridge on Moravian Drive,  
0.2 mile downstream from North Branch, and 0.5  
mile west of Mount Clemens.
- 8US23 Belle River at Memphis, lat. 42° 54' 03", long.  
83° 46' 09", in NW 1/4 of SE 1/4 sec. 35, T.6N.,  
R.14E., St. Clair County, on right bank, at  
downstream side of bridge on State Highway 19 at  
Memphis.

- 8US25 Pine River, SE 1/4 sec. 27, T.5N., R.16E., at Vine Road, 3.0 miles northwest of St. Clair. (Base-flow discharge measurements only -- made on 5/15/67 and 7/31/67).
- 8US27 Black River near Port Huron, section 2, T.6N., R.16E., at highway bridge 6 miles west of Port Huron. (Continuous records for water years 1933-1943 only).
- 8US28 Mill Creek (Black River) near Avoca, lat. 43° 03' 16", long. 82° 44' 05", in NW 1/4 sec. 8, T.7N., R.15E., St. Clair County, on left bank at downstream side of bridge on Bricker Road, 0.2 miles upstream from Gleason Drain, and 2.3 miles west of Avoca.
- 8US29 Black River near Fargo, lat. 42° 05' 32", long. 82° 37' 05", in NW 1/4 sec. 32, T.8N., R.16E., St. Clair County, on left bank 20 ft. downstream from bridge on Norman Road, 2.1 miles east of Fargo, 5.3 miles upstream from Mill Creek, and 12 miles northwest of Port Huron.
- 8US34 Mill Creek (Black River) near Abbottsford, lat. 43° 03', long. 82° 37', NW 1/4 sec. 17, T.7N., R.16E., on downstream side of highway bridge, 1 mile upstream from mouth and 2 miles northeast of Abbottsford. (Continuous records for May 1947-September 1964 (discontinued)).

ONTARIO MINISTRY OF THE ENVIRONMENT (OME)

- 103 Headwater of Niagara River; 9 sampling points in Lake Erie ranging from Ontario to New York shores as follows (ft.): 200, 1000, 2000, 3500, 5500, 6500, 8500, 10,000, 11,500
- 403 Detroit River, Range DT 3.9; sampling points from west shore across range as follows (ft.):  
 403.1 - 200, 1500, 2500, 3500  
 403.2 - 4500, 5500, 6500  
 403.3 - 7500, 8500  
 403.4 - 9500, 11,500, 13,500  
 403.5 - 14,500, 15,000, 16,200  
 403.6 - 16,500, 17,000  
 403.7 - 17,500, 18,500, 19,000, 19,300
- 4040 Detroit River, at Range DT 12.0W.; sampling points from west shore across range as follows (ft.):  
 122, 322, 490, 670, 750, 888
- 4043 Detroit River, Range DT 13.12; sampling points from west across range as follows (ft.):  
 200, 400, 600



- 4088 Detroit River, Range DT 30.7E.; sampling points from Peach Island across range as follows (ft.):  
4088.1 - 100, 300  
4088.2 - 500, 700, 850, 900, 980
- 4089 Detroit River, Range DT 30.8W.; sampling points from U.S. shore across range as follows (ft.):  
4089.1 - 20, 100, 200, 300  
4089.2 - 400, 500, 700  
4089.3 - 1000  
4089.4 - 2000, 2500  
4089.5 - 2750, 3000, 3250, 3500, 3750, 4000, 4250
- 606 St. Clair River, Range SR 13.7; sampling points from west shore across range as follows (ft.):  
100, 700, 1000, 1400, 1900
- 6025 St. Clair River, Range SR 39.0; sampling points from west shore across range as follows (ft.):  
100, 700, 1000, 1400, 1900
- 901 Big Creek, at Highway No. 18, County of Essex.  
Upstream mileage - 3.3
- 902 Canard River, at Highway No. 18. Upstream mileage - 0.5
- 903 Turkey Creek, at Highway No. 18. Upstream mileage - 0.2
- 904 Little River, at Riverside Drive, Windsor.  
Upstream mileage - 0.1
- 905 Parent Drain, at Riverside Drive west of Tecumseh.  
Within 200 ft. of mouth
- 906 Manning Drain, at Riverside Drive, Riverside.  
Within 200 ft. of mouth
- 907 Pike Creek, at Tecumseh Road, Highway No. 39.  
Upstream mileage - 0.3
- 908 Puce River, at Highway No. 39. Upstream mileage - 0.4
- 909 Belle River, at CNR bridge, Village of Belle River.  
Upstream mileage - 0.2
- 9010 Ruscom River, at Tecumseh Road, Rochester Township.  
Upstream mileage - 0.6
- 9012 Thames River, at Lake St. Clair, 4 sampling points:  
- Right side (facing upstream), - Center, top (i.e. surface), - Center, bottom (i.e. depth sample), - Left side (facing upstream). Upstream mileage - 0.1

- 9016 Sydenham River, at Highway No. 40, 2 sampling points:  
- Right side (facing upstream), - Left side (facing upstream). Upstream milage - 2.8
- 9020 Talford Creek, at Highway No. 40. Upstream mileage - 0.2
- 9021 County Road Ditch, at Polymer Corporation, City of Sarnia, 2 sampling points:  
- Left side (facing upstream), - Right side (facing upstream). Within 100 ft. of mouth.

MICHIGAN WATER RESOURCES COMMISSION (MWRC)

- 3M14 Lake Erie at Detroit River mouth, Detroit Light to Pte. Mouillee Light Range, 3 sampling points west of Detroit Light (ft.):  
11,500, 12,500, 13,000
- 3M15 Lake Erie at Detroit River mouth, Detroit Light to Pte. Mouillee Light Range, 2 sampling points west of Detroit Light (ft.):  
9500, 10,700
- 3M16 Lake Erie at Detroit River mouth, Detroit Light to Pte. Mouillee Light Range, 4 sampling points west of Detroit Light (ft.):  
5000, 6000, 7000, 8000
- 3M18 Lake Erie at Detroit River mouth, Detroit Light to Pte. Mouillee Light Range, 3 sampling points west of Detroit Light (ft.):  
1000, 2000, 3580
- 4M4 Range DT 3.9, Maple Beach to Bar Pt., 2500 ft. east of Maple Beach
- 4M6 Range DT 3.9, 3500 ft. E. of Maple Beach
- 4M8 Range DT 3.9, 4500 ft. E. of Maple Beach
- 4M9 Range DT 3.9, 5500 ft. E. of Maple Beach
- 4M11 Range DT 3.9, 6500 ft. E. of Maple Beach
- 4M12 Range DT 3.9, 7500 ft. E. of Maple Beach
- 4M13 Range DT 3.9, 9500 ft. E. of Maple Beach
- 4M15 Range DT 3.9, 11,500 ft. E. of Maple Beach
- 4M16 Range DT 3.9, 13,500 ft. E. of Maple Beach
- 4M18 Range DT 3.9, 2 sampling points E. of Maple Beach (ft.):  
14,500, 15,000
- 4M19 Range DT 3.9, 2 sampling points E. of Maple Beach (ft.):  
16,200, 16,500

4M21 Range DT 3.9, 17,500 ft. E. of Maple Beach

4M23 Range DT 3.9, 3 sampling points E. of Maple Beach (ft.):  
18,500, 19,000, 19,300

4M28 Range DT 8.7, Gross Isle Co. Br., 80 ft. from west shore

4M29 Range DT 8.7, 280 ft. from west shore

4M31 Range DT 8.7, 480 ft. from west shore

4M32 Range DT 8.7, 680 ft. from west shore

4M34 Range DT 8.7, 2 sampling points from west shore (ft.):  
980, 1240

4M41 Range DT 12.0, Gross Isle Toll Br., 670 ft. from west shore

4M42 Range DT 12.0, 880 ft. from west shore

4M45 Range DT 14.6, Wyandotte City Power Plant, 5 sampling points from west shore (ft.):  
20, 100, 200, 300, 400

4M47 Range DT 14.6, 2 sampling points from west shore (ft.):  
800, 1000

4M48 Range DT 14.6, 2000 ft. from west shore

4M50 Range DT 14.6, 3000 ft. from west shore

4M51 Range DT 14.6, 4000 ft. from west shore

4M52 Range DT 14.6, 6000 ft. from west shore

4M58 Range DT 19.0, 4 sampling points from west shore (ft.):  
100, 200, 300, 400

4M59 Range DT 19.0, 800 ft. from west shore

4M61 Range DT 19.0, 1000 ft. from west shore

4M62 Range DT 19.0, 1500 ft. from west shore

4M63 Range DT 19.0, 2000 ft. from west shore

4M64 Range DT 19.0, 3 sampling points from west shore (ft.):  
2200, 2300, 2400

4M66 Range DT 19.0, 2500 ft. from west shore

4M68 Range DT 20.6, Det. Edison Dock - Canad. Salt, 2 sampling points from west shore (ft.):  
200, 400

4M69 Range DT 20.6, 2 sampling points from west shore (ft.):  
600, 700

4M71	Range DT 20.6, 1000 ft. from west shore
4M72	Range DT 20.6, 1500 ft. from west shore
4M73	Range DT 20.6, 1800 ft. from west shore
4M75	Range DT 20.6, 2 sampling points from west shore (ft.): 2000, 2300
4M76	Range DT 20.6, 2 sampling points from west shore (ft.): 5, 50
4M84	Range DT 30.7, Peach Is. to Edgewater Inn, 500 ft. from west shore
4M85	Range DT 30.7, 850 ft. from west shore
4M87	Range DT 30.7, 980 ft. from west shore
4M91	Range DT 30.8, Windmill Pte. to N. Peach Is., 2 sampling points from U.S. shore (ft.): 20, 100
4M92	Range DT 30.8, 300 ft. from west shore
4M94	Range DT 30.8, 500 ft. from west shore
4M95	Range DT 30.8, 1000 ft. from west shore
4M97	Range DT 30.8, 2 sampling points from west shore (ft.): 2000, 2500
4M98	Range DT 9.3, Grosse Ile (foot of Ferry Road - High School) to Canadian shoreline (Yellow House), 200 ft. from Grosse Ile shore
4M99	Range DT 9.3, 500 ft. from Grosse Ile shore
4M100	Range DT 9.3, 1200 ft. from Grosse Ile shore
4M101	Range DT 9.3, 3000 ft. from Grosse Ile shore
4M102	Range DT 9.3, 4000 ft. from Grosse Ile shore
4M103	Range DT 9.3, 5000 ft. from Grosse Ile shore
4M104	Range DT 9.3, 5800 ft. from Grosse Ile shore
4M105	Range DT 12.0, Grosse Ile Toll Br., 122 ft. from west shore
4M106	Range DT 12.0, 490 ft. from west shore
4M107	Range DT 17.0, Fighting Island to N. end of La Salle Oil Dock, 100 ft. from Fighting Island shore
4M108	Range DT 17.0, 400 ft. from Fighting Island shore
4M109	Range DT 17.0, 900 ft. from Fighting Island shore

- 4M110 Range DT 30.7, 100 ft. from west shore
- 4M117 Range DT 14.6, 1400 ft. from west shore, at water works intake
- 8M3 River Raisin at Monroe, County of Monroe. At Renault Import Dock, lat.  $41^{\circ} 54' 02''$ , long.  $83^{\circ} 21' 16''$ . Upstream mileage - 1.13.
- 8M6 Huron River below Rockwood, County of Monroe. At River Road Bridge, 3.5 miles southeast of Rockwood, lat.  $42^{\circ} 02' 43''$ , long.  $83^{\circ} 12' 51''$ . Upstream mileage - 1.70
- 8M10 River Rouge (New Channel, at mouth) at River Rouge, County of Wayne. At Detroit, Toledo, and Ironton Railroad Bridge south of Zug Island, lat.  $42^{\circ} 16' 41''$ , long.  $83^{\circ} 06' 56''$ . 3 Sampling points from west shore (ft.): 50, 100, 150; also 4th point, midstream, sampled on different day each month from other 3 points. Upstream mileage - 0.36
- 8M18 Clinton River below Mt. Clemens, County of Macomb. At Bridgeview Road bridge, 3 mi. east of Mt. Clemens, lat.  $42^{\circ} 35' 47''$ , long.  $82^{\circ} 49' 35''$ . Upstream mileage - 3.50.
- 8M22 Belle River at Marine City, County of St. Clair. At Bridge Street bridge in Marine City, lat.  $42^{\circ} 42' 48''$ , long.  $82^{\circ} 29' 48''$ . Upstream mileage - 0.48.
- 8M24 Pine River at St. Clair, County of St. Clair. At State Highway M-29 bridge in St. Clair, lat.  $42^{\circ} 49' 12''$ , long.  $82^{\circ} 29' 10''$ . Upstream mileage - 0.05.
- 8M26 Black River at Port Huron, County of St. Clair. At end of catwalk, downstream from Chesapeake and Ohio Railroad bridge at end of Clyde St., lat.  $42^{\circ} 58' 25''$ , long.  $82^{\circ} 25' 13''$ . Upstream mileage - 0.08.

#### ENVIRONMENTAL PROTECTION AGENCY (EPA)

- 1E4 Lake Erie at Buffalo, lat.  $42^{\circ} 52' 48''$ , long.  $78^{\circ} 54' 44''$
- 1E11 Lake Erie at Buffalo Water Supply Intake, lat.  $42^{\circ} 52' 46''$ , long.  $78^{\circ} 54' 45''$
- 2E1 Maumee River, at C. & O. Railroad Docks, 3 sampling points:  
 - lat.  $41^{\circ} 41' 46''$ , long.  $83^{\circ} 27' 39''$   
 - lat.  $41^{\circ} 41' 46''$ , long.  $83^{\circ} 27' 38''$   
 - lat.  $41^{\circ} 41' 46''$ , long.  $83^{\circ} 21' 39''$



3E7	lat. 41° 51' 52", long. 83° 19' 44"
3E8	lat. 41° 53' 17", long. 83° 19' 20"
3E9	lat. 41° 54' 04", long. 83° 19' 11"
3E10	lat. 41° 55' 08", long. 83° 17' 10"
3E12	lat. 41° 57' 00", long. 83° 13' 54"
3E13	lat. 41° 58' 26", long. 83° 12' 30"
3E17	lat. 42° 00' 03", long. 83° 09' 19"
3E32	lat. 41° 50' 16", long. 83° 20' 55"
3E33	lat. 41° 47' 09", long. 83° 23' 19"
4E5	Range DT 3.9, 2500 ft. from west shore, lat. 42° 03' 13", long. 83° 10' 41"
4E7	Range DT 3.9, 4500 ft. from west shore, lat. 42° 03' 14", long. 83° 10' 15"
4E10	Range DT 3.9, 6500 ft. from west shore, lat. 42° 03' 14", long. 83° 09' 48"
4E14	Range DT 3.9, 11500 ft. from west shore, lat. 42° 03' 15", long. 83° 08' 35"
4E17	Range DT 3.9, 15000 ft. from west shore, lat. 42° 03' 15", long. 83° 07' 50"
4E20	Range DT 3.9, 18500 ft. from west shore, lat. 42° 03' 17", long. 83° 07' 20"
4E22	Range DT 3.9, 19300 ft. from west shore, lat. 42° 03' 17", long. 83° 07' 00"
4E27	Range DT 8.7 W, 80 ft. from west shore, lat. 42° 07' 38", long. 83° 10' 40"
4E33	Range DT 8.7 W, 2 sampling points from west shore: - 680 ft., lat. 42° 07' 37", long. 83° 10' 32" - 1240 ft., lat. 42° 07' 37", long. 83° 10' 25"
4E35	Range DT 9.3 E, 200 ft., lat. 42° 08' 10", long. 83° 08' 15"
4E36	Range DT 9.3 E, 1200 ft., lat. 42° 08' 10", long. 83° 08' 01"
4E37	Range DT 9.3 E, 4000 ft., lat. 42° 08' 12", long. 83° 07' 24"
4E38	Range DT 9.3 E, 5000 ft., lat. 42° 08' 12", long. 83° 07' 11"
4E39	Range DT 9.3 E, 5600 ft., lat. 42° 08' 12", long. 83° 07' 03"

4E44 Range DT 14.6 W., 3 sampling points:  
 - 20 ft., lat. 42° 12' 28", long. 83° 08' 41"  
 - 100 ft., lat. 42° 12' 28", long. 83° 08' 40"  
 - 400 ft., lat. 42° 12' 28", long. 83° 08' 37"

4E46 Range DT 14.6 W., 1000 ft., lat. 42° 12' 29",  
 long. 83° 08' 29"

4E49 Range DT 14.6 W., 3000 ft., lat. 42° 12' 30",  
 long. 83° 08' 02"

4E53 Range DT 17.0 E., 400 ft., lat. 42° 14' 15",  
 long. 83° 06' 33"

4E54 Range DT 17.0 E., 900 ft., lat. 42° 14' 16",  
 long. 83° 06' 26"

4E55 Range DT 17.4 W., 2 sampling points:  
 - 100 ft., lat. 42° 14' 50", long. 83° 07' 44"  
 - 400 ft., lat. 42° 14' 49", long. 83° 07' 41"

4E56 Range DT 17.4 W., 1600 ft., lat. 42° 14' 42",  
 long. 83° 07' 28"

4E57 Range DT 19.0, 2 sampling points:  
 - 100 ft., lat. 42° 16' 02", long. 83° 06' 47"  
 - 400 ft., lat. 42° 16' 00", long. 83° 06' 43"

4E60 Range DT 19.0, 1000 ft., lat. 42° 15' 57",  
 long. 83° 06' 36"

4E65 Range DT 19.0, 2500 ft., lat. 42° 15' 53", long.  
 83° 06' 17"

4E67 Range DT 20.6, 3 sampling points:  
 - 5 ft., lat. 42° 17' 33", long. 83° 05' 57"  
 - 50 ft., lat. 42° 17' 33", long. 83° 05' 57"  
 - 400 ft., lat. 42° 17' 31", long. 83° 05' 53"

4E70 Range DT 20.6, 1000 ft., lat. 42° 17' 28", long.  
 83° 05' 46"

4E74 Range DT 20.6, 2 sampling points:  
 - 2000 ft., lat. 42° 17' 26", long. 83° 05' 34"  
 - 2300 ft., lat. 42° 17' 24", long. 83° 05' 32"

4E83 Range DT 30.7 E., 500 ft., lat. 42° 20' 32",  
 long. 82° 55' 46"

4E86 Range DT 30.7 E., 980 ft., lat. 42° 20' 27",  
 long. 82° 55' 48"

4E90 Range DT 30.8 W., 100 ft., lat. 42° 21' 27",  
 long. 82° 55' 47"

4E93 Range DT 30.8 W., 500 ft., lat. 42° 21' 23",  
 long. 82° 55' 45"

4E96 Range DT 30.8 W., 2500 ft., lat. 42° 21' 05",  
 long. 82° 55' 35"

5E1	lat. 42° 20' 00", long. 82° 45' 00"
5E2	lat. 42° 24' 35", long. 82° 50' 02"
5E3	lat. 42° 25' 00", long. 82° 40' 00"
5E4	lat. 42° 24' 05", long. 82° 53' 03"
5E5	lat. 42° 27' 37", long. 82° 52' 18"
5E6	lat. 42° 30' 00", long. 82° 50' 00"
5E7	lat. 42° 30' 33", long. 82° 42' 00"
5E8	lat. 42° 32' 38", long. 82° 51' 08"
5E9	lat. 42° 34' 08", long. 82° 49' 02"
5E10	lat. 42° 35' 00", long. 82° 45' 00"
5E11	lat. 42° 35' 42", long. 82° 46' 40"
5E12	lat. 42° 37' 30", long. 82° 45' 00"
5E13	lat. 42° 40' 37", long. 82° 43' 57"
5E14	lat. 42° 40' 23", long. 82° 38' 58"
6E1	Range SR 10.0 N., 100 ft., lat. 42° 36' 41", long. 82° 32' 09"
6E2	Range SR 10.0 N., 900 ft., lat. 42° 36' 33", long. 82° 32' 04"
6E3	Range SR 10.0 S., 100 ft., lat. 42° 36' 28", long. 82° 31' 31"
6E4	Range SR 10.0 S., 900 ft., lat. 42° 36' 22", long. 82° 31' 25"
6E7	Range SR 13.7, 100 ft., lat. 42° 39' 31", long. 82° 30' 51"
6E8	Range SR 13.7, 1000 ft., lat. 42° 39' 30", long. 82° 30' 38"
6E9	Range SR 13.7, 1900 ft., lat. 42° 39' 27", long. 82° 30' 22"
6E15	Range SR 26.7, 100 ft., lat. 42° 50' 29", long. 82° 28' 31"
6E16	Range SR 26.7, 800 ft., lat. 42° 50' 27", long. 82° 28' 22"
6E17	Range SR 26.7, 1900 ft., lat. 42° 50' 25", long. 82° 28' 08"
6E22	Range SR 39.0, 100 ft., lat. 43° 00' 22", long. 82° 25' 19"
6E23	Range SR 39.0, 800 ft., lat. 43° 00' 16", long. 82° 25' 10"

6E24	Range SR 39.0, 1500 ft., lat. 43° 00' 12", long. 82° 25' 01"
7E1	lat. 43° 05' 30", long. 82° 27' 07"
7E2	lat. 43° 05' 30", long. 82° 24' 54"
7E3	lat. 43° 05' 30", long. 82° 21' 31"
7E4	lat. 43° 05' 30", long. 82° 18' 10"
7E5	lat. 43° 05' 30", long. 82° 14' 48"
7E6	lat. 43° 05' 30", long. 82° 11' 25"
7E7	lat. 43° 49' 30", long. 82° 26' 00"
7E8	lat. 43° 47' 45", long. 82° 11' 30"
7E9	lat. 43° 47' 00", long. 82° 04' 30"
7E10	lat. 43° 46' 00", long. 81° 56' 15"
7E11	lat. 43° 44' 45", long. 81° 45' 45"

POLLUTION CONTROL AGENCY, TOLEDO (TPCA)

2T2	Maumee River at Chesapeake and Ohio Coal Dock
2T3	Maumee River at Toledo Terminal Railroad Bridge (West side)
2T4	Maumee River at Sports Arena

INSTITUTE OF WATER RESEARCH, M.S.U. (IWR)

3I1	lat. 41° 47' 36", long. 83° 22' 54"
3I2	lat. 41° 50' 48", long. 83° 20' 48"
3I3	lat. 41° 52' 12", long. 83° 19' 48"
3I4	lat. 41° 53' 36", long. 83° 18' 54"
3I5	lat. 41° 54' 30", long. 83° 18' 12"
3I6	lat. 41° 55' 42", long. 83° 17' 24"

JOINT USGS - EPA

4U-E30	Detroit River at Trenton, lat. 42° 07' 37", long. 83° 10' 35", in SE 1/4 sec. 19, T.4S., R.11E., Wayne County, 480 ft. from right bank at Grosse Isle Parkway Bridge over Trenton Channel, 11 miles south of Detroit.
4U-E82	Detroit River at Detroit, at Detroit municipal raw-water intake for Water Works Park Filtration Plant.

4U-E98 Detroit River at Gibraltar, lat. 42° 05' 29", long. 83° 11' 06", in NW 1/4 sec. 6, T.5S., R.11E., Wayne County, at intake to EPA (FWQA) automatic monitor location 300 feet out from bank of Trenton Channel, 13.5 miles south of Detroit. (Station superseded by 4U-E30, which is approx. 2.5 miles upstream).

6U-E20 St. Clair River at Port Huron, lat. 42° 59' 19", long. 82° 25' 29", in SE 1/4 sec. 3, T.6N., R.17E., St. Clair County, 125 ft. out from right bank at Port Huron municipal raw-water intake at Pine Grove Park.

DETROIT METROPOLITAN WATER DEPARTMENT (DMWD)

4D111 Detroit River. In 1500 ft. long by 400 ft. wide lagoon, northeast extremity of Belle Isle - raw-water intake for Water Works Park Filtration Plant.

4D112 Detroit River. Off west shore of Fighting Island, off eastern edge of Fighting Island Channel, lat. 42° 13' 36.5", long. 83° 07' 42.1" - raw-water intake for Southwest Filtration Plant.

NATIONAL OCEAN SURVEY - LAKE SURVEY CENTER (NOAA)

1N5 Buffalo - Station No. 3020:  
Buffalo, Erie County, New York, lat. 42° 52' 39", long. 78° 53' 27", at west end of South Pier at Coast Guard Station.

1N6 Black Rock Canal - Station No. 3017:  
Buffalo, Erie County, New York, lat. 42° 56' 02", long. 78° 54' 27", at Buffalo District, foot of Bridge St., near Lower Gate on West Lock Wall.

1N7 Ontario St. - Station No. 3117:  
Buffalo, Erie County, New York, lat. 42° 56' 35", long. 78° 54' 37", on the Niagara River, at the foot of Ontario St., on the upstream corner of the Industrial Molasses Corp. dock.

1N8 Niagara Intake - Station No. 3012:  
Niagara Falls, Niagara County, New York, lat. 43° 04' 40", long. 79° 00' 50", on river bank, near foot of Iroquois Avenue, near most easterly intake gate.

1N9 American Falls - Station No. 3010:  
Niagara Falls, Niagara County, New York, lat. 43° 04' 55", long. 79° 03' 45", on U.S. shore, opposite Head of Goat Island.



- 1N10 Ashland Avenue - Station No. 3007:  
Niagara Falls, Niagara County, New York, lat. 43° 06' 00", long. 79° 03' 39", at Niagara Falls Sewage Treatment Plant at foot of Ashland Avenue.
- 3N31 Fermi Power Plant - Station No. 3090:  
At Fermi Power Plant, near Stony Point, Monroe County, Michigan, lat. 41° 57' 35", long. 83° 15' 30", on shore near Fermi Power Plant water intake.
- 4N113 Gibraltar - Station No. 4020:  
Gibraltar, Wayne County, Michigan, lat. 42° 05' 30", long. 83° 11' 10", at the foot of Grandview Avenue in Gibraltar.
- 4N114 Wyandotte - Station No. 4030:  
Wyandotte, Wayne County, Michigan, lat. 42° 12' 12", long. 83° 08' 50", near foot of Oak Street in back of building at 3025 Van Alstyne Blvd.
- 4N115 Fort Wayne - Station No. 4036:  
Detroit, Wayne County, Michigan, lat. 42° 17' 53", long. 83° 05' 34", at the foot of Livernois Ave., on Detroit District Corps of Engineers Boatyard dock.
- 4N116 Windmill Point - Station No. 4049:  
Detroit, Wayne County, Michigan, lat. 42° 21' 26", long. 82° 55' 45", near foot of Alter Road, on Public Health Service grounds, near Windmill Point Light.
- 6N26 Algonac - Station No. 4070:  
Algonac, St. Clair County, Michigan, lat. 42° 37' 15", long. 82° 31' 40", on downriver corner of the Waterworks Dock at foot of Green St.
- 6N27 St. Clair State Police - Station No 4080:  
St. Clair, St. Clair County, Michigan, lat. 42° 48' 55", long. 82° 29' 09" on the downstream side of the St. Clair State Police Post property.
- 6N28 Marysville - Station No. 4084:  
Marysville, St. Clair County, Michigan, lat. 42° 54' 22", long. 82° 27' 55", on Filtration Plant dock at the foot of Huron Blvd.
- 6N29 Dry Dock - Station No 4087:  
Port Huron, St. Clair County, Michigan, lat. 42° 56' 40", long. 82° 26' 45", at foot of Grant Place, in front of 3300 Military Ave.
- 6N30 Mouth of Black River (MBR) - Station No. 4090:  
Port Huron, St. Clair County, Michigan, lat. 42° 58' 25", long. 82° 25' 12", on north side of Black River, on downriver end of Gulf Refining Co. dock in rear of Paton Oil Co.

6N31

Dunn Paper - Station No. 4096:

Port Huron, St. Clair County, Michigan, lat.  $43^{\circ}$   
 $00' 15''$ , long.  $82^{\circ} 25' 20''$ , in angle of Dunn  
Paper Co. dock.

6N32

Fort Gratiot - Station No. 4098:

Port Huron, St. Clair County, Michigan, lat.  $43^{\circ}$   
 $00' 23''$ , long.  $82^{\circ} 25' 21''$ , at Fort Gratiot Coast  
Guard Station.

7N12

Lakeport - Station No. 5002:

Lakeport, St. Clair County, Michigan, lat.  $43^{\circ}$   
 $08' 30''$ , long.  $82^{\circ} 29' 35''$ , at the foot of Harris  
Road in Burtchville Community Park.

### *Hydrology:*

Using the 1969-1971 data base we determined the contribution of stream flow to the hydraulics of the impact zone. Although these estimates have been made before it was necessary for us to go into much more detail since the precise estimates of materials loading needed for our study were highly dependent on accurate flow information. This procedure involved studying daily flow rates over the three year study period for the 21 tributaries in the impact zone. Monthly and yearly means were then calculated from this base. Figure 9 summarizes this information and Tables 29-32 present more detail on stream flow during the study period.

Six of the seven southeastern Michigan streams considered were gaged during the study period. The gaging stations are upstream in order to minimize the backwater effect of the Great Lakes and connecting waterways. Therefore, monthly flow values were adjusted to reflect the greater drainage areas at the mouths; factors (ratio of respective drainage areas at mouths and at gages) ranged from 1.03 to 1.52. The Pine River is ungaged, therefore streamflow was synthesized from runoff in adjacent basins (Belle River and Mill Creek), recognizing that the Pine River has lower low-flows and higher high-flows than adjoining basins because of its clay soil. In southwestern Ontario, only the Thames and Sydenham Rivers are gaged. Flow in the County Road Ditch is sustained primarily by industrial discharge. Flows in the remaining nine ungaged streams were estimated, using empirical procedures based on correlations with adjacent gaged streams; the factors considered were monthly flow of a gaged stream in an approximately homogeneous region, the ratio of drainage areas, and the ratio of estimated mean annual runoff. These estimates were adjusted for any excessive regional precipitation. The same formula was used to estimate the monthly flows at the mouth of the Thames River, considering the gaged area above Thamesville and below Byron as one entity, and the ungaged area below Thamesville as another entity. The monthly flows of the Sydenham River at its mouth were estimated, using

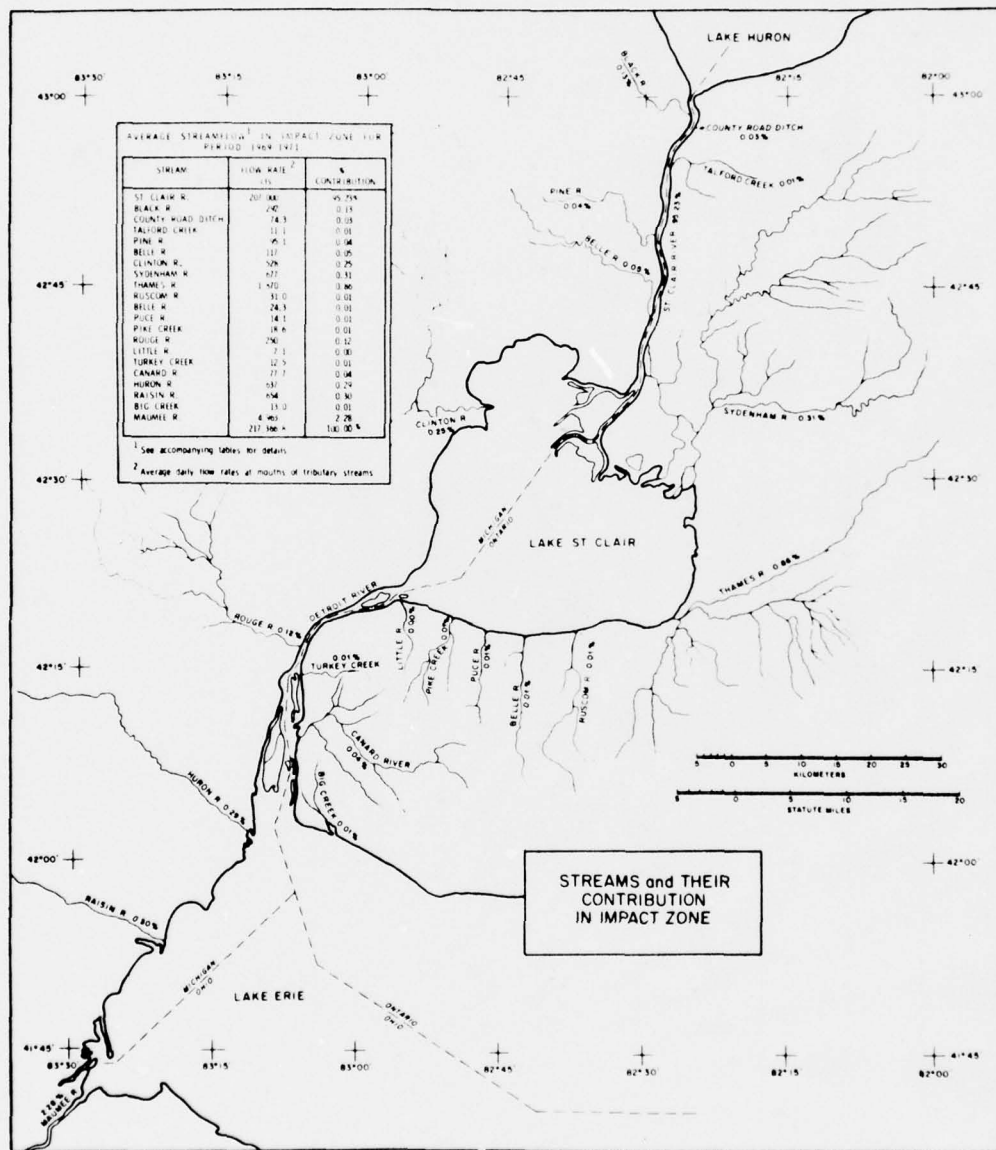


Figure 9. Stream discharge into the impact zone.

Table 29. Volumetric flow<sup>1</sup> through the impact zone, 1969-1971.<sup>2</sup>

SAMPLING REGION	FLOW RATE, cfs				% FLOW				Period Of Record
	1969	1970	1971	Period Of Record <sup>3</sup>	1969	1970	1971	1971	
St. Clair River	204,000	201,000	215,000	187,600 <sup>4</sup>	93.2	96.2	101.4	92.9	
Southeastern Michigan Tributary Streams	3,251	2,299	2,167	2,651 <sup>5</sup>	1.5	1.1	1.0	1.3	
Southwestern Ontario Tributary Streams	3,646	2,391	2,453	2,890 <sup>6</sup>	1.7	1.1	1.2	1.4	
Detroit River	210,000	205,000	219,000	188,600 <sup>4</sup>	-	-	-	-	
Maumee River	6,297	4,826	3,765	4,867 <sup>7</sup>	2.9	2.3	1.8	2.4	
Niagara River	219,000	209,000	212,000	202,000 <sup>4</sup>	100	100	100	100	

<sup>1</sup> Average daily flow rates

<sup>2</sup> Calendar years

<sup>3</sup> Through December, 1971

<sup>4</sup> 112 years (1860-1971)

<sup>5</sup> Various years-of-record of six major streams; seventh major stream (Pine River) is ungaged - flow synthesized



Table 29. (con't.)

<sup>6</sup>Twelve streams considered are unengaged (flow synthesized) - except for the Sydenham and Thames Rivers; the figure given is the summation of the arithmetic averages for 1969-1971 for the unengaged ten streams and the Sydenham River (only 4 years of record), plus the average flow for the period-of-record of the Thames River (16 years of record)

<sup>7</sup>46 years (1922-1935, 1940-1971)

Table 30. Great Lakes tributary streamflow<sup>1</sup> at mouths, sampling regions 8 and 9, 1969-1971.<sup>2</sup>

Stream	Flow Rate, cfs				% Streamflow Contribution							
	1969	1970	1971	Period Of Record <sup>4</sup>	Regionally			Overall				
					1969	1970	1971	Period Of Record	1969	1970	1971	Period Of Record
Discharging to St. Clair River:												
Michigan:	471.4	432.9	606	575.1	84.4	83.4	87.9	87.0	6.8	9.2	13.1	10.4
Black R.	268	232	375	376.5	48.0	44.7	54.4	56.8	3.9	5.0	8.1	6.8
Pine R.	89.4	86.9	109	95.1 <sup>6</sup>	16.0	16.7	15.8	14.4	1.3	1.8	2.4	1.7
Belle R.	114	114	122	104.7	20.4	22.0	17.7	15.8	1.6	2.4	2.6	1.9
Ontario:	87.1	86.2	83.0	85.4	15.6	16.6	12.1	13.0	1.3	1.9	1.8	1.5
County Rd.												
Ditch <sup>3</sup>	74.3	74.3	74.3	74.3 <sup>6</sup>	13.3	14.3	10.8	11.3	1.1	1.6	1.6	1.3
Talford C.	12.8	11.9	8.7	11.1 <sup>6</sup>	2.3	2.3	1.3	1.7	0.2	0.3	0.2	0.2
Sub-Total	558.5	519.1	689.0	660.5	100.	100.	100.	100.	8.1	11.1	14.9	11.9
-----												
Discharging to Lake St. Clair:												
Michigan:	667	483	435	507	16.4	18.0	16.0	15.8	9.7	10.3	9.4	9.2
Clinton R.	667	483	435	507 <sup>8</sup>	16.4	18.0	16.0	15.8	9.7	10.3	9.4	9.2
Ontario:	3,409.5	2,207.5	2,286.8	2,694	83.6	82.0	84.0	84.2	49.4	47.0	49.5	48.6
Sydenham R.	810	630	590	677 <sup>9</sup>	19.9	23.4	21.7	21.2	11.7	13.4	12.8	12.2
Thames R.	2,480	1,500	1,630	1,929 <sup>10</sup>	60.8	55.7	59.8	60.2	35.9	32.0	35.3	34.8

Table 30. (con't.)

Ruscom R.	42.1	27.3	23.5	31.0 <sup>6</sup>	1.0	1.0	0.9	1.0	0.6	0.6	0.5	0.6
Belle R.	33.0	21.4	18.5	24.3 <sup>6</sup>	0.8	0.8	0.7	0.8	0.5	0.5	0.4	0.4
Puce R.	19.1	12.4	10.7	14.1 <sup>6</sup>	0.5	0.5	0.4	0.4	0.3	0.2	0.2	0.3
Pike C.	25.3	16.4	14.1	18.6 <sup>6</sup>	0.6	0.6	0.5	0.6	0.4	0.3	0.3	0.3
Sub-Total	4,076.5	2,690.5	2,721.8	3,201	100.	100.	100.	100.	59.1	57.3	58.9	57.8

## Discharging to Detroit River:

Michigan:	351	201	198	286	72.6	70.1	72.8	74.5	5.1	4.3	4.3	5.2
Rouge R.	351	201	198	286 <sup>11</sup>	72.6	70.1	72.8	74.5	5.1	4.3	4.3	5.2
Ontario:	132.2	85.6	73.9	97.3 <sup>6</sup>	27.4	29.9	27.2	25.5	1.9	1.8	1.6	1.7
Little R.	9.6	6.2	5.4	7.1 <sup>6</sup>	2.0	2.2	2.0	1.9	1.1	0.1	0.1	0.1
Turkey C.	17.0	11.0	9.5	12.5 <sup>6</sup>	3.5	3.8	3.5	3.3	0.3	0.2	0.2	0.2
Canard R.	105.6	68.4	59.0	77.7 <sup>6</sup>	21.9	23.9	21.7	20.3	1.5	1.5	1.3	1.4
Sub-Total	483.2	286.6	271.9	383.3	100.	100.	100.	100.	7.0	6.1	5.9	6.9

## Discharging to Western Lake Erie:

Michigan:	1,762	1,182	928	1,283	99.0	99.0	98.9	99.0	25.5	25.2	20.1	23.2
Huron R.	839	594	477	596 <sup>12</sup>	47.1	49.7	50.8	46.0	12.1	12.7	10.3	10.8
Raisin R.	923	588	451	687 <sup>13</sup>	51.9	49.3	48.1	53.0	13.4	12.5	9.8	12.4
Ontario:	17.7	11.5	9.9	13.0	1.0	1.0	1.1	1.0	0.3	0.3	0.2	0.2
Big C.	17.7	11.5	9.9	13.0 <sup>6</sup>	1.0	1.0	1.1	1.0	0.3	0.3	0.2	0.2
Sub-Total	1,779.7	1,193.5	937.9	1,296	100.	100.	100.	100.	25.8	25.5	20.3	23.4

## Overall-

Total	6,897.9	4,689.7	4,620.6	5,540.8	100.	100.	100.	100.	100.	100.	100.	100.
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Table 30. (con't.)

- <sup>1</sup> Average daily flow rates
- <sup>2</sup> Calendar years
- <sup>3</sup> Flow sustained by industrial wastewater discharges and storm runoff
- <sup>4</sup> Through December, 1971
- <sup>5</sup> 24 years (1948-1971)
- <sup>6</sup> Ungaged streams; figures given are arithmetic averages of synthesized flow for 1969-1971
- <sup>7</sup> 9 years (1963-1971)
- <sup>8</sup> 37 years (1935-1971)
- <sup>9</sup> 4 years (1968-1971); however, 1968 flow not used -- too high - would be misleading for such a short period-of-record
- <sup>10</sup> 16 years (1956-1971)
- <sup>11</sup> 24 years (1948-1971)
- <sup>12</sup> 23 years (1949-1971)
- <sup>13</sup> 34 years (1938-1971)

**Table 31.** Streamflow percentage contribution of Southeastern Michigan streams to Great Lakes and connecting waterways, 1969-1971.

Stream	Year	% Streamflow Contribution:				Overall
		St. Clair River	Lake St. Clair	Detroit River	Western Lake Erie	
Black River	1969	56.8	-	-	-	8.2
	1970	53.6	-	-	-	10.1
	1971	61.9	-	-	-	17.3
	Period Of					
	Record	75.4	-	-	-	14.2
Pine River	1969	19.0	-	-	-	2.8
	1970	20.1	-	-	-	3.8
	1971	18.0	-	-	-	5.0
	Period Of					
	Record	16.5				3.6
Belle River	1969	24.2	-	-	-	3.5
	1970	26.3	-	-	-	5.0
	1971	20.1	-	-	-	5.6
	Period Of					
	Record	18.1	-	-	-	3.9
Clinton River	1969	-	100	-	-	20.5
	1970	-	100	-	-	21.0
	1971	-	100	-	-	20.1
	Period Of					
	Record	-	100	-	-	19.1
Rouge River	1969	-	-	100	-	10.8
	1970	-	-	100	-	8.7
	1971	-	-	100	-	9.2
	Period Of					
	Record	-	-	100	-	10.8



Table 31. (con't.)

Huron River	1969	-	-	-	47.6	25.8
	1970	-	-	-	50.3	25.8
	1971	-	-	-	51.4	22.0
	Period Of Record	-	-	-	46.4	22.5
<hr/>						
Raisin River	1969	-	-	-	52.4	28.4
	1970	-	-	-	49.7	25.6
	1971	-	-	-	48.6	20.8
	Period Of Record	-	-	-	53.6	25.9
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		<hr/>	<hr/>	<hr/>	<hr/>	<hr/>
		100	100	100	100	100
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Table 32. Streamflow percentage contribution of streams entering  
Lake St. Clair and Western Lake Erie, 1969-1971.

	% Streamflow Contribution			
Stream	1969	1970	1971	Period of Record
----- Lake St. Clair: -----				
Clinton River (Mich.)	0.32	0.24	0.20	0.27
Ontario:	1.64	1.09	1.05	1.41
Sydenham River	0.39	0.31	0.27	0.35
Thames River	1.19	0.74	0.75	1.01
Ruscom River	0.02	0.01	0.01	0.02
Belle River	0.02	0.01	0.01	0.01
Puce River	0.01	0.01	0.00	0.01
Pike Creek	0.01	0.01	0.01	0.01
St. Clair River	<u>98.04</u>	<u>98.67</u>	<u>98.75</u>	<u>98.32</u>
	100.	100.	100.	100.
----- Western Lake Erie: -----				
Michigan:	0.80	0.56	0.41	0.66
Huron River	0.38	0.28	0.21	0.31
Raisin River	0.42	0.28	0.20	0.35
Big Creek (Ont.)	0.01	0.01	0.01	0.01
Maumee River (Ohio)	2.89	2.28	1.68	2.50
Detroit River	<u>96.30</u>	<u>97.15</u>	<u>97.90</u>	<u>96.83</u>
	100.	100.	100.	100.

the same procedure as for the gaged southeastern Michigan streams, the factor used was 1.486. Similarly, the Maumee River's monthly mean flows for each of the years 1969-1971 were estimated, using the factor 1.044.

Using the outflow of the Niagara River as 100% it can be seen that approximately 93% of this flow originates from Lake Huron. The largest single tributary in the impact zone (excluding the Detroit and St. Clair Rivers) is the Maumee River which contributes slightly over 2% to the Niagara discharge. Southeast Michigan tributaries and Southwestern Ontario tributaries are quite similar in total discharge, each contributing a little more than 1%. It should be noted that not all river systems tributary to the Niagara River area have been included. Watersheds east of the Maumee River and the Detroit River tributary to the central and eastern portions of Lake Erie appear to have little hydrological influence on this portion of the basin.

#### *Materials Loading:*

Figure 10 shows the locations of stream mouth and connecting waterway sampling stations that were utilized in our loading calculations. The key on the facing page for Figure 10 outlines the specific sampling stations at each stream mouth location and station detail for selected Great Lakes connecting waterway locations are provided in Table 33. Location of sampling stations on the Great Lakes proper are shown in Figure 11.

At the onset of our investigation we examined a total of 28 water quality parameters. Approximate loadings of each of these constituents was given in our Phase I report. For Phase II we concentrated on the more biologically important parameters limiting our efforts to analysis of nutrient inputs. These included phosphorus, nitrogen and carbon compounds. Very early in our assessments we concluded that of the nutrient materials, phosphorus represented the most critical element in governing the trophic status of Lake Erie. This is not to say that the current trophic condition of the lake system is limited by phosphorus, but rather

for the lake to achieve a desirable quality it would have to become phosphorus limited. Thus, inputs of BOD and nitrogen assume a secondary role. Further discussion of nutrient limitations for the entire Lake Erie basin are given later in this report in Sections IV and VI.

Our decision to concentrate on phosphorus loadings was further supported by the waste treatment technologies with their high removal efficiencies that are being prepared for southeastern Michigan. Initial examination of the environmental impacts of the proposed wastewater management alternatives indicated that phosphorus removal from municipal and industrial sources would drastically reduce the total phosphorus input to the Lake Erie system. The magnitude of this reduction appeared to result in a loading that would generate a condition bordering that of a biological phosphorus limitation for the western basin. For this reason we felt it was essential to characterize phosphorus dynamics very carefully to achieve optimum resolution between the impacts of the different management alternatives.

The calculations for phosphorus loadings were relatively straight forward involving a multiplication of stream discharge times the concentration of the element. This process is unfortunately complicated by many factors.

Loadings at locations in Great Lakes connecting waterways were computed as the products of mean annual discharges and concentrations. In the Detroit River, streamflow distribution was considered, as were stations encompassed by the percentage-of-flow divisions. Average concentrations of two or more stations in a particular flow distribution zone were weighted in proportion to the number of samples at each of the agencies' stations. The final loading at a location was computed as the weighted average of agencies' loadings; proportional to the total number of samples taken by each agency at all its stations at the particular location. The degree of accuracy and precision of a particular agencies' sampling methods and laboratory procedures was not evaluated. However, when station concentrations for each of the years 1969-1971 were arranged in

Figure 10. Comprehensive map of sampling regions 1-9 showing locations of computed loadings for 1969-1971.

KEY	
Location	Description
D39	Detroit R. - Mouth, at IJC Range DT 3.9
D307	Detroit R. - at IJC Range DT 30.7E
D308	Detroit R. - at IJC Range DT 30.8W
D307-8	Detroit R. - Headwater, at IJC Ranges DT 30.7E and DT 30.8W
M	Maumee R., Ohio - Mouth (2E1, 2T2-4, 2US9)
MBe	Belle R., Michigan - Mouth (8M22, 8US23)
MB1	Black R., Michigan - Mouth (8M26, 8US27-29)
MC	Clinton R., Michigan - Mouth (8M18, 8US19)
MH	Huron R., Michigan - Mouth (8M6, 8US7, 8US9)
MP	Pine R., Michigan - Mouth (8M24, 8US23, 8US25, 8US28)
MRa	Raisin R., Michigan - Mouth (8M3, 8US4)
MRO	Rouge R., Michigan - Mouth (8M10, 8US15-17)
N	Niagara R. - Headwater, at Exit from Lake Erie
OBe	Belle R., Ontario - Mouth (909, 9CA17-19, 9CA22, 9CA24-25)
OBi	Big Creek, Ontario - Mouth (901, 9CA17-19, 9CA22, 9CA24-25)
OCa	Canard R., Ontario - Mouth (902, 9CA17-19, 9CA22, 9CA24-25)
OCo	County Road Ditch, Ontario - Mouth (9021)
OL	Little R., Ontario - Mouth (904, 9CA17-19, 9CA22, 9CA24-25)
OPi	Pike Creek, Ontario - Mouth (907, 9CA17-19, 9CA22, 9CA24-25)
OPu	Puce R., Ontario - Mouth (908, 9CA17-19, 9CA22, 9CA24-25)
OR	Ruscom R., Ontario - Mouth (9010, 9CA17-19, 9CA22, 9CA24-25)
OS	Sydenham R., Ontario - Mouth (9016, 9CA17, 9CA18)
OTa	Talford Creek, Ontario - Mouth (9020, 9CA19)
OTh	Thames R., Ontario - Mouth (9012, 9CA14, 9CA23)
OTu	Turkey Creek, Ontario - Mouth (903, 9CA17-19, 9CA22, 9CA24-25)
S137	St. Clair R. - Mouth, at IJC Range SR 13.7
S390	St. Clair R. - Headwater at IJC Range SR 39.0



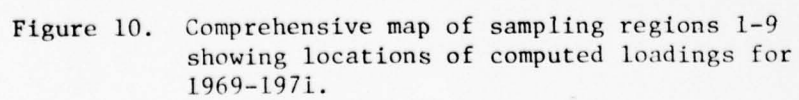


Table 33. Summary of selected locations sampled for total phosphorus (as P) in Great Lakes connecting waterways, sampling regions 1, 4 and 6, 1969-1971.

Location	Stations	Number of		
		Stations	Sampling Points	Samples
N	1E4	1	1	158
	103	<u>1</u>	<u>9</u>	<u>96</u>
		2	10	254
D39	4M4, 4M9, 4M12, 4M13, 4M15, 4M18, 4M19, 4M23	8	12	156
	4E5, 4E7, 4E10, 4E14, 4E17, 4E20, 4E22	7	7	63
	403.1, 403.2, 403.3, 403.4, 403.5, 403.6, 403.7	<u>7</u>	<u>21</u>	<u>286</u>
		22	40	505
D307	4M110, 4M84, 4M87	3	3	35
	4E83, 4E86	2	2	18
	4088.1, 4088.2	<u>2</u>	<u>7</u>	<u>106</u>
		7	12	159
D308	4M91, 4M92, 4M94, 4M95, 4M97	5	7	82
	4E90, 4E93, 4E96	3	3	27
	4089.1, 4089.2, 4089.3, 4089.4, 4089.5	<u>5</u>	<u>17</u>	<u>250</u>
		13	27	359
S137	6E7, 6E8, 6E9	3	3	45
	606	<u>1</u>	<u>5</u>	<u>130</u>
		4	8	175
S390	6E22, 6E23, 6E24	3	3	47
	6025	<u>1</u>	<u>5</u>	<u>81</u>
		4	8	128

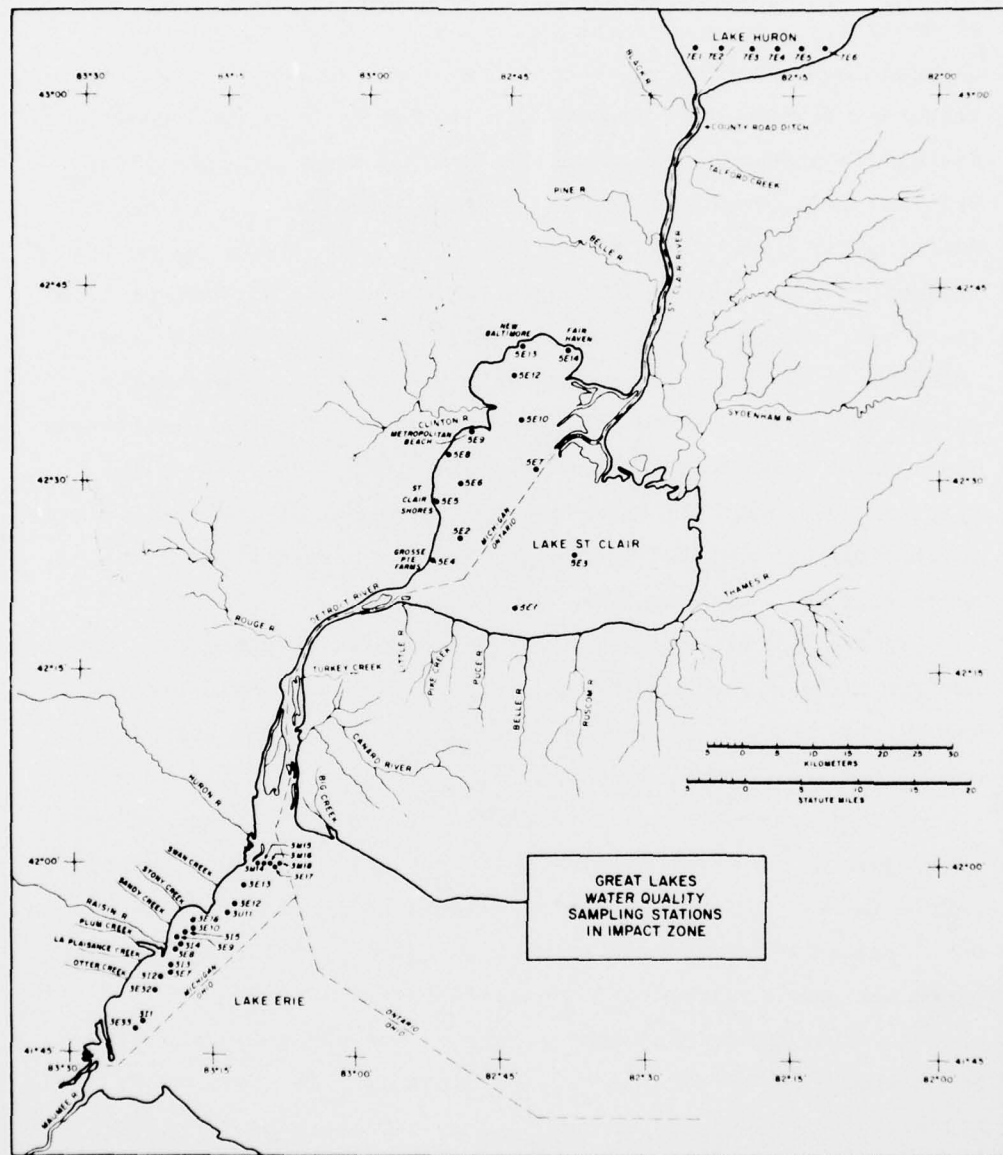


Figure 11. Great Lakes water quality sampling stations.

ascending order, as a preliminary step, an occasional, unusual value was easily spotted and deleted from subsequent computations, obviating a loading which would have been misleading. Variation of concentrations with depth was not significant for loading computations. Loadings at locations at the mouths of Great Lakes tributary streams were computed as the products of mean monthly discharges and graphically-derived monthly mean concentrations. Original concentrations at each station were plotted, usually monthly, for each of the years 1969-1971. The graphical method attenuates fluctuations in concentration; especially abrupt highs and lows. Judgement was used in deriving concentrations from among three stations in the Maumee River, best representative of actual conditions at the mouth; one station (2T3) was immediately upstream of the sewage treatment plant outfall; the other two stations (2E1 and 2T2) were downstream, almost in the same location in the lake; prevailing wind direction was a factor in affecting concentrations when sampling.

In short, basic hydrological information is quite well developed for the study area, records of water quality are not. Missing values were common and composite samples, reflecting water quality over a period of time, were severely lacking.

During periods of intense rain a stream will discharge very large quantities of phosphorus. If, for some reason, a "grab" sample is not taken during this period a calculated loading may be a drastic underestimate of what is actually entering the lake. Since the probability of a predetermined sampling routine encountering these infrequent storms is low and the probability of encountering relatively "dry" periods is high there is a tendency for calculated loadings to be underestimates. These factors were carefully considered in our calculations but fortunately, the more important sampling stations such as the Detroit River gave much more reliable estimates due to extensive surveillance activities in this area (See Table 34). We were also fortunate in being able to cross check calculated loading by using water quality values obtained at different times by different agencies.

Our final loading values were weighted averages of data obtained by most agencies sampling at a particular location.

The contribution of materials from non-point sources were estimated based on values reported in the literature and studies on Michigan watersheds conducted by the Institute of Water Research and other agencies in the state. In the case of phosphorus, it has been clearly shown that releases from all sources will eventually translocate from the watershed into the receiving lake system over an annual period. Moreover, since phosphorus distributes itself between the solution and particulate phases of a stream system it probably represents the extreme in the duration of residency within the stream network. Thus, it can be concluded that an estimate for materials balance should be taken on the basis of cumulative annual values. We did so and will present these values for each year of the three year period of 1969-1971.

Table 35 summarizes phosphorus loadings into the western Lake Erie impact zone. Over 6 million pounds of phosphorus per year enter Lake Erie via the discharge of Lake Huron into the St. Clair River. By the time the Lake Huron flow enters the western basin of Lake Erie at the mouth of the Detroit River it has picked up an additional 24 million pounds per year of phosphorus. It is interesting to note that of the 24 million pounds per year only slightly over 2 million pounds per year enter by Michigan and Ontario tributary streams. Detail on individual stream inputs of phosphorus to the connecting waterway stream is shown in Table 36. The relative percentage contribution of the Michigan tributary streams to Lake Erie is outlined in Table 37. As can be seen here the input of the Clinton River accounts for nearly half of the phosphorus input carried by Michigan streams. This loading, however, is equaled by the input from the Thames River in Ontario. The single largest phosphorus loading by a tributary stream into the western basin (excluding the Detroit River) is that from the Maumee River; over 5 million pounds per year. The total relative contribution of phosphorus to the western basin is summarized in Table 38.



Table 34. Total phosphorus (as P) concentrations at stations at selected locations in Great Lakes connecting waterways, sampling regions 1, 4 and 6, 1969-1971.

Location	Station	No. of Sampling Points	Months Sampled <sup>1</sup>			No. of Samples			Arith. Mean Concentration (Mg/L)		
			1969	1970	1971	1969	1970	1971	1969	1970	1971
N	1E4	1	1-12	1-12	1-12	51	52	55	0.025	0.031	0.020
	1O3	9	5-8,10	7,9	6-9	44	18	34	0.035	0.042	0.028
D39	4M4	1	4,5,8,9,11	5,7,9,10	5,6,7,9,11	5	5	6	0.538	0.296	0.228
	4M9	1	4,5,8,9,11	5,7,9,10	5,6,7,9,11	5	5	6	0.208	0.218	0.163
	4M12	1	4,5,8,9,11	5,7,9,10	5,6,7,9,11	5	5	6	0.142	0.143	0.122
	4M13	1	4,5,8,9,11	5,7,9,10	5,6,7,9,11	5	5	6	0.104	0.112	0.067
	4M15	1	4,5,8,9,11	5,7,9,10	5,6,7,9,11	5	5	6	0.062	0.062	0.038
	4M18	2	4,5,8,9,11	5,7,9,10	5,6,7,9,11	5	5	12	0.042	0.068	0.031
	4M19	2	4,5,8,9,11	5,7,9,10	5,6,7,9,11	5	5	12	0.052	0.054	0.028
	4M23	3	4,5,8,9,11	5,7,9,10	5,6,7,9,11	10 45	10 45	12 66	0.059	0.057	0.038
	4E5	1	5-10	5,6,8	---	6	3	---	0.502	0.203	---
	4E7	1	5-10	5,6,8	---	6	3	---	0.182	0.157	---
	4E10	1	5-10	5,6,8	---	6	3	---	0.130	0.100	---
	4E14	1	5-10	5,6,8	---	6	3	---	0.047	0.037	---
403.1 403.2 403.3	4E17	1	5-10	5,6,8	---	6	3	---	0.067	0.040	---
	4E20	1	5-10	5,6,8	---	6	3	---	0.045	0.037	---
	4E22	1	5-10	5,6,8	---	6 42	3 21	---	0.140	0.037	---
								0			
	403.1	4	4,6-10	6,7,12	5-10	24	12	14	0.338	0.278	0.173
	403.2	3	4,6-10	6,7,12	5-10	16	12	6	0.130	0.118	0.087
	403.3	2	4,6-10	6,7,12	5-10	11	6	8	0.065	0.049	0.073

Table 34. (con't.)

403.4	3	4,6-10	6,7,12	5-10	17	9	16	0.046	0.042	0.047
403.5	3	4,6-10	6,7,12	5-10	6	3	46	0.035	0.047	0.025
403.6	2	4,6-10	6,7,12	5-10	11	6	15	0.031	0.042	0.023
403.7	4	4,6-10	6,7,12	5-10	24	12	12	0.045	0.047	0.028
					109	60	117			
<hr/>										
D307	4M110	1	---	6,7,9,11	--	--	4	---	---	0.020
	4M84	1	4,5,7-9	5,6,8,9,11	5	5	6	0.088	0.066	0.032
	4M87	1	4,5,7-9	5,6,8,9,11	4	5	6	0.098	0.034	0.037
					9	10	16			
	4E83	1	5-10	---	7	2	--	0.043	0.015	---
	4E86	1	5-10	---	7	2	--	0.041	0.015	---
					14	4				
	4088.1	2	4,6,7,9,10	5,7,11,12	11	8	11	0.033	0.033	0.017
	4088.2	5	4,6,7,9,10	5,7,11,12	30	19	27	0.053	0.034	0.023
					41	27	38			
<hr/>										
D308	4M91	2	4,5,7-9	5,6,8,9,11	5	5	10	0.188	0.090	0.044
	4M92	1	4,5,7-9	5,6,8,9,11	5	5	5	0.104	0.072	0.036
	4M94	1	4,5,7-9	5,6,8,9,11	5	5	5	0.078	0.046	0.028
	4M95	1	4,5,7-9	5,6,8,9,11	5	5	6	0.066	0.042	0.020
	4M97	2	4,5,7-9	5,6,8,9,11	5	5	6	0.052	0.040	0.022
					25	25	32			
	4E90	1	5-10	---	7	2	--	0.070	0.040	---
	4E93	1	5-10	---	7	2	--	0.051	0.030	---
	4E96	1	5-10	---	7	2	--	0.021	0.010	---
					21	6				

Table 34. (con't.)

4089.1	4	4,6,7,9,10	5,7,11,12	5-10	24	16	30	0.062	0.055	0.039
4089.2	3	4,6,7,9,10	5,7,11,12	5-10	17	12	10	0.029	0.026	0.022
4089.3	1	4,6,7,9,10	5,7,11,12	5-10	6	4	10	0.018	0.036	0.017
4089.4	2	4,6,7,9,10	5,7,11,12	5-10	12	8	16	0.017	0.028	0.016
4089.5	7	4,6,7,9,10	5,7,11,12	5-10	41	28	16	0.032	0.031	0.016
					100	68	82			
SL37	6E7	1	4-10	---	9	--	6	0.027	---	0.012
	6E8	1	4-10	---	9	--	6	0.027	---	0.013
	6E9	1	4-10	---	9	--	6	0.022	---	0.012
					27		18			
606	5	4,6,7,8,10	5,7,11,12	5,6,8,9,10	25	20	85	0.012	0.025	0.016
S390	6E22	1	4-6,8-10	5,6,8	8	3	6	0.026	0.010	0.017
	6E23	1	4-6,8-10	5,6,8	7	3	5	0.016	0.011	0.007
	6E24	1	4-6,8-10	5,6,8	6	3	6	0.012	0.008	0.018
					21	9	17			
6025	5	4,6-8,10	4,7,11,12	5,6,8-10	15	19	47	0.012	0.014	0.014

<sup>1</sup>1 (January), ..., 12 (December)

Table 35. Selected total phosphorus (as P) loadings, 1969-1971.

Source	Loading, lb			Average
	1969	1970	1971	
Lake Huron, at headwater of St. Clair River (S390 <sup>1</sup> )	6,456,049	-	5,922,800	6,189,425
All sources upstream of St. Clair River mouth (S137 <sup>1</sup> )	7,526,482	-	6,473,185	6,999,834
All sources above headwater of Detroit River (D307-8 <sup>1</sup> )	14,204,065	13,287,448	8,997,777	12,163,097
All sources upstream of Detroit River mouth (D39 <sup>1</sup> )	37,033,528	30,655,260	23,808,771	30,499,186
Southeastern Michigan tributary streams <sup>2</sup>	2,542,227	1,854,987	1,259,156	1,885,456
Southwestern Ontario tributary streams <sup>3</sup>	1,574,375	1,234,834	1,315,504	1,374,904
Maumee River (M <sup>4</sup> )	7,365,268	5,234,781	3,082,479	5,227,509
Lake Erie, at headwater of Niagara River (N <sup>1</sup> )	12,769,077	13,912,031	9,617,979	12,099,696

<sup>1</sup> Loadings based on concentrations as determined by ONE and EPA

<sup>2</sup> Loadings based on concentrations as determined by MTRC

Table 35. (Con't.)

- <sup>3</sup> Loadings based on concentrations as determined by OME
- <sup>4</sup> Loadings based on concentrations as determined by EPA and TPCA



In short, it is quite obvious that sources other than tributary streams account for the bulk of the phosphorus discharged into the western basin of Lake Erie. Discussion of the non-stream sources will be presented later in the report in the section on Ecological Impacts (Section VI).

Table 36. Great Lakes tributary streams total phosphorus (as P) loading at mouths, sampling regions 8 and 9, 1969-1971.

Stream	Loading <sup>1</sup> , lb			Ave. % Loading Contribution		
	1969	1970	1971	Ave.	Regionally	Overall
Discharging to St. Clair River:						
Michigan:						
Black R. (MB1)	61,840	36,903	55,342	112,740	92.09	3.46
Pine R. (MP)	25,266	18,181	27,267	51,362	41.96	1.58
Belle R. (MBc)	47,748	32,652	33,022	23,571	19.25	.72
				37,807	30.88	1.16
Ontario:						
County Road Ditch (OCo)	6,940	6,722	5,444	9,690	7.91	.30
Talford Creek (OTa)	6,831	988	2,144	6,369	5.20	.20
				3,321	2.71	.10
Sub-Total				122,430	100.	3.76
Discharging to Lake St. Clair:						
Michigan:						
Clinton R. (MC)	1,161,741	889,307	431,451	827,500	40.79	25.38
				827,500	40.79	25.38
Ontario:						
Sydenham R. (OS)	277,491	170,301	185,185	1,201,394	59.21	36.85
Thames R. (OTh)	1,024,247	795,923	986,243	210,993	10.40	6.47
Ruscom R. (OR)	18,361	7,760	11,212	935,471	46.11	28.69
Belle R. (OBe)	15,651	14,567	16,801	12,444	0.61	.38
Puce R. (OPu)	12,154	12,478	18,412	15,673	0.77	.48
				14,348	0.71	.44

Table 36. (Con't.)

Pike Creek (OPI)	13,527	17,542	6,327	12,465	0.61	.38
Sub-Total				2,028,894	100.	
Discharging to Detroit River:						
Michigan:						
Rouge R. (MRO)	266,453	74,840	36,183	125,825	44.46	3.86
				125,825	44.46	3.86
Ontario:						
Little R. (OL)	13,570	30,752	13,855	157,172	55.54	4.82
Turkey Creek (OTu)	104,661	70,314	37,423	19,391	6.85	.60
Canard R. (OCa)	72,167	100,788	27,992	70,799	25.02	2.17
				66,982	23.67	2.05
Sub-Total				282,997	100.	8.68
Discharging to western Lake Erie:						
Michigan:						
Huron R. (MH)	465,695	446,271	355,280	819,391	99.20	25.13
Raisin R. (MRA)	513,484	356,833	320,611	422,415	51.14	12.95
				396,976	48.06	12.17
Ontario:						
Big Creek (OBi)	8,775	6,699	4,466	6,647	0.80	.20
				6,647	0.80	.20
Sub-Total				826,038	100.	25.33
OVERALL-TOTAL				3,260,359		100.

<sup>1</sup> Obtained from summation of monthly loadings.

Table 37. Total phosphorus (as P) percentage loading contribution of southeastern Michigan streams to Great Lakes and connecting waterways, 1969-1971.

Stream	Ave. % Loading Contribution, 1969-1971				
	St. Clair River	Lake St. Clair	Detroit River	Western Lake Erie	Overall
Black R. (MB1)	45.56	--	--	--	2.72
Pine R. (MP)	20.91	--	--	--	1.25
Belle R. (MBe)	33.53	--	--	--	2.01
Clinton R. (MC)	--	100	--	--	43.90
Rouge R. (MRo)	--	--	100	--	6.67
Huron R. (MH)	--	--	--	51.55	22.40
Raisin R. (MRa)	--	--	--	48.45	21.05
	100.	100	100	100.	100.

Table 38. Total phosphorus (as P) percentage loading contribution of streams entering Lake St. Clair and western Lake Erie, 1969-1971.

Stream	Ave. % Loading Contribution, 1969-1971
LAKE ST. CLAIR:	
Clinton River (Mich.) (MC)	9.16
Ontario:	13.31
Sydenham River (OS)	2.34
Thames River (OTh)	10.36
Ruscom River (OR)	0.14
Belle River (OBe)	0.17
Puce River (OPu)	0.16
Pike Creek (OPi)	0.14
St. Clair River (S137)	<u>77.53</u>
	100.
WESTERN LAKE ERIE:	
Michigan:	2.24
Huron River (MH)	1.15
Raisin River (MRa)	1.09
Big Creek (Ont.) (OBi)	0.02
Maumee River (Ohio) (M)	14.30
Detroit River (D39)	<u>83.44</u>
	100.



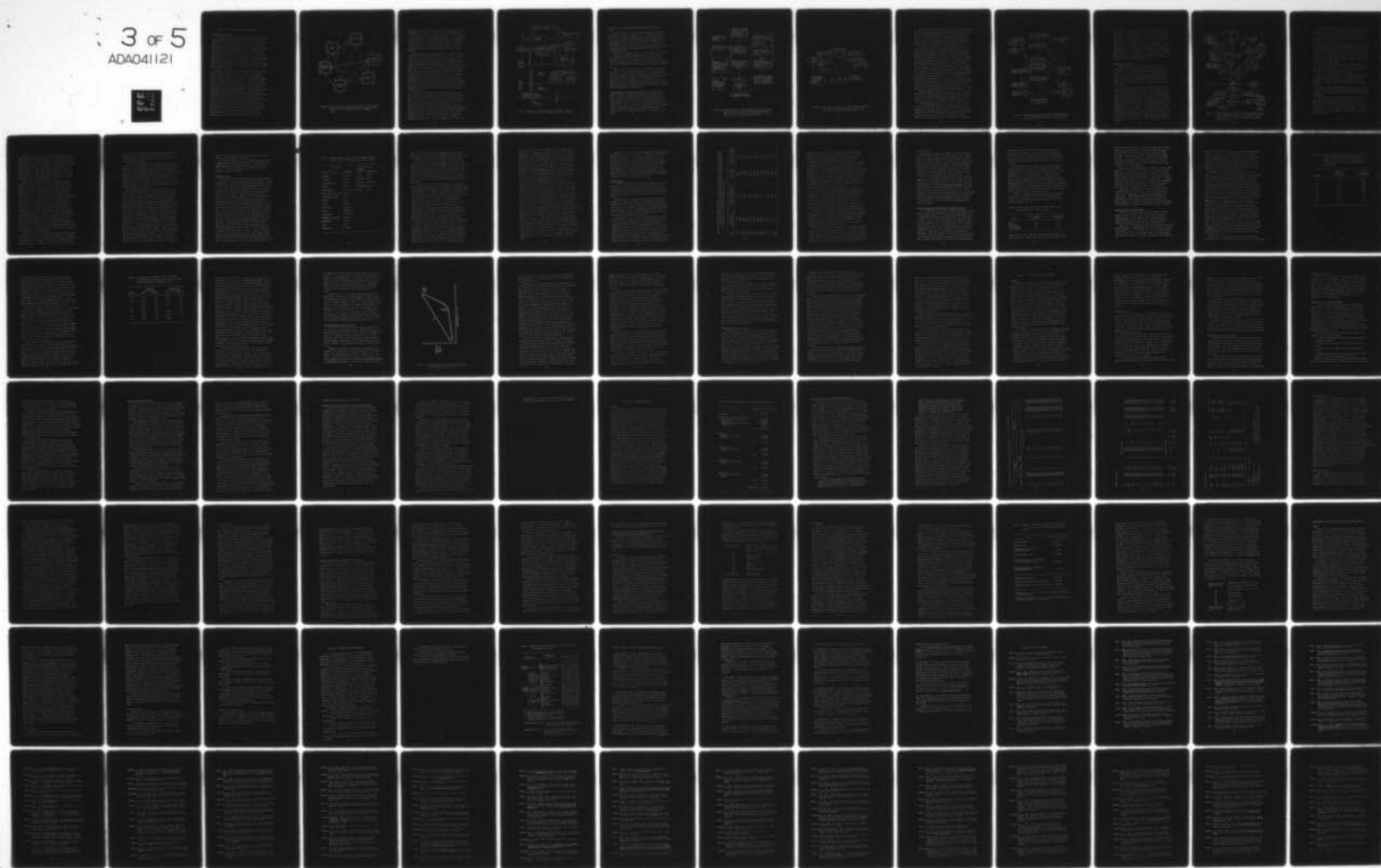
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ECOLOGICAL ASSESSMENTS FOR WASTEWATER MANAGEMENT IN SOUTHEAST--ETC(U)  
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#### SECTION IV - GENERAL ECOLOGICAL CONSIDERATIONS

##### Terrestrial Ecosystems

A terrestrial ecosystem can be simply defined as a biotic and abiotic realm that contains structure and information. Strict delineation of such a system is not possible, but certain characteristics and functions may be ascribed in order to facilitate understanding of biological complexes. In the following discussions we hope to provide the reader with the necessary perspective to appreciate the complexity of ecosystem function. Unlike many of the physical sciences the study of ecosystems is in a very early, primarily philosophical, phase of inquiry. In fact, most hypotheses concerning ecosystem function were generated over the last century. A number of precepts, however, are generally accepted among ecologists and these will be utilized in the assessments of wastewater management strategies.

Typically, an ecosystem may be divided into subsets of interacting components (Figure 12). Physical and chemical components such as soils and climatic relationships form the foundation that determines which state a biological system will tend towards. Biological components of trophic structure and diversity provide pathways for assimilation and processing of energy, nutrients, and essential elements. Storage and accumulation paths provide an information content that in turn influences and guides successional development and evolutionary adaptation. Adaptive mechanisms possessed by biological species, assure behavioral and physiological responses, and thus provide a relatively sensitive cybernetic aspect to the entire system. That is, within the limits of species' tolerance and genetic ability to adapt to new conditions, the biological realm is a dynamic entity, constantly adjusting to new conditions and pressures. In this manner, life on earth has been maintained in spite of several drastic shifts in the earth's general physiognomy. There is little reason to suspect that organisms will suddenly cease to be adaptive, but evolutionary

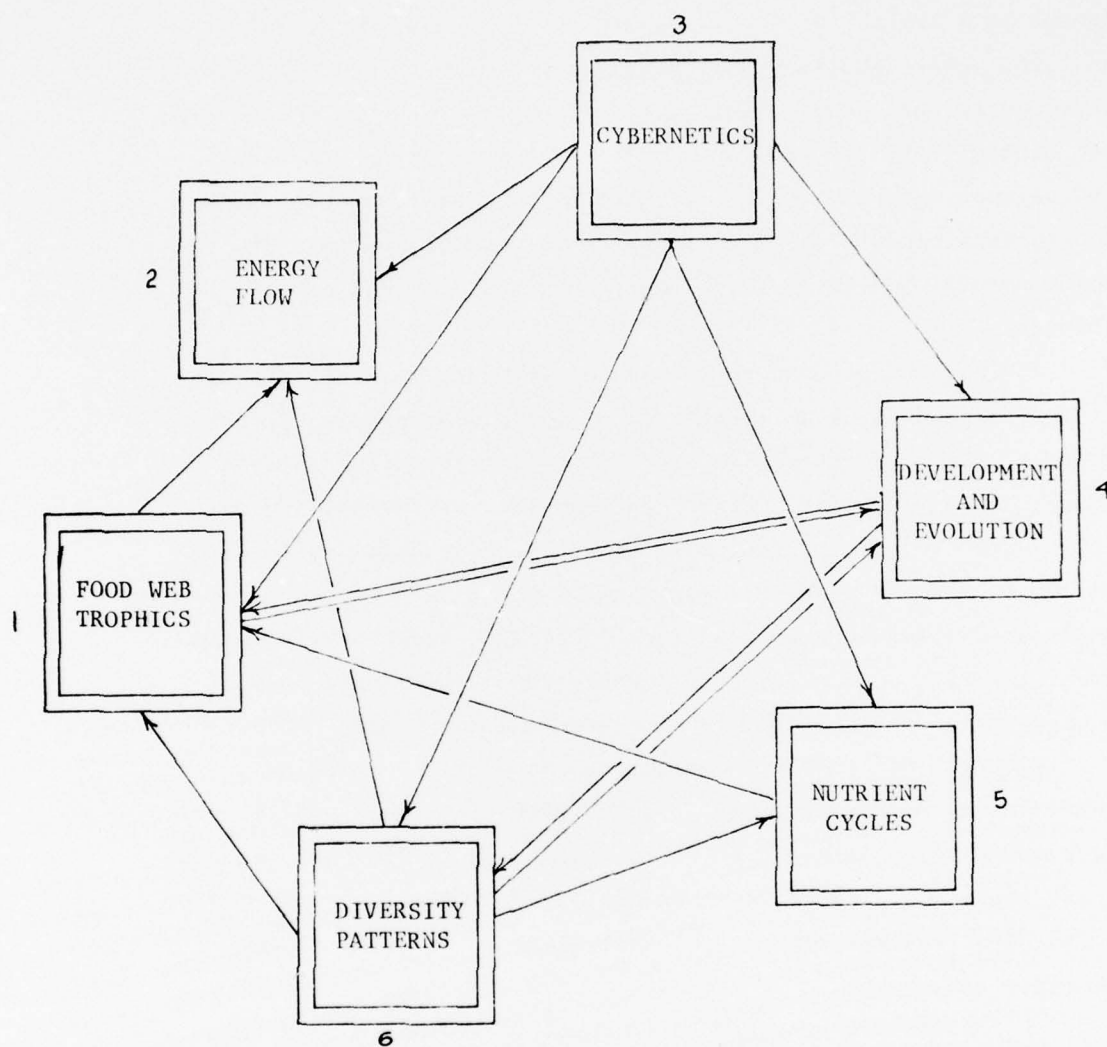


Figure 12. Process pattern in a terrestrial ecosystem. Interactions between these six components determines ultimate system stability. See text for discussion of components 1, 3, 4 and 6.

history makes clear the fact that great realms of animal and plant groups have arisen only to be superceded by a subsequent wave of forms better adapted to a changing condition. Thus, if mankind is to perpetuate, two factors remain paramount. We must maintain our biologic ability to adapt to inevitably changing conditions, and we must maintain a store of biological support systems that are in turn capable of adjusting both to naturally occurring environmental shifts (e.g. weather trends) and to man caused changes.

The maintenance and processing of water as it moves through biological and physical systems is of prime importance. Normally, water in the form of rain is provided to a terrestrial system, which then divides into various pathways for optimum biological utilization (Figure 13). Historically, the hydrologic cycle is a balanced cycle (excluding ice age buildup and decline), but recently, man's activities have shunted ever-increasing amounts by pumping and irrigation into paths that make it impossible for a typical ecosystem to provide complete refinement and recharge. Resultant contamination and runoff increases the rate of system change, subsequently resulting in deletion of many biological species that lack the ability to adapt to rapidly changing conditions. Agricultural practices promote these conditions artificially, with added derangement contributed by toxic materials designed to further simplify the system.

The ability of a terrestrial system to process and utilize water ultimately depends upon conditions both within and outside of our control. Soil structure and content is largely beyond man's control. Percolation rates, depth of saturation zones, and structural ability to support given vegetative types are critical, and will ultimately determine a site's suitability as a receiver of effluent discharge. Adsorption properties, complexing mechanisms, retention and release, and erosion sensitivities involve physical properties that are largely beyond man's control at other than superficial levels. Such factors can, however, be maximized through wise management of vegetative communities, provided that

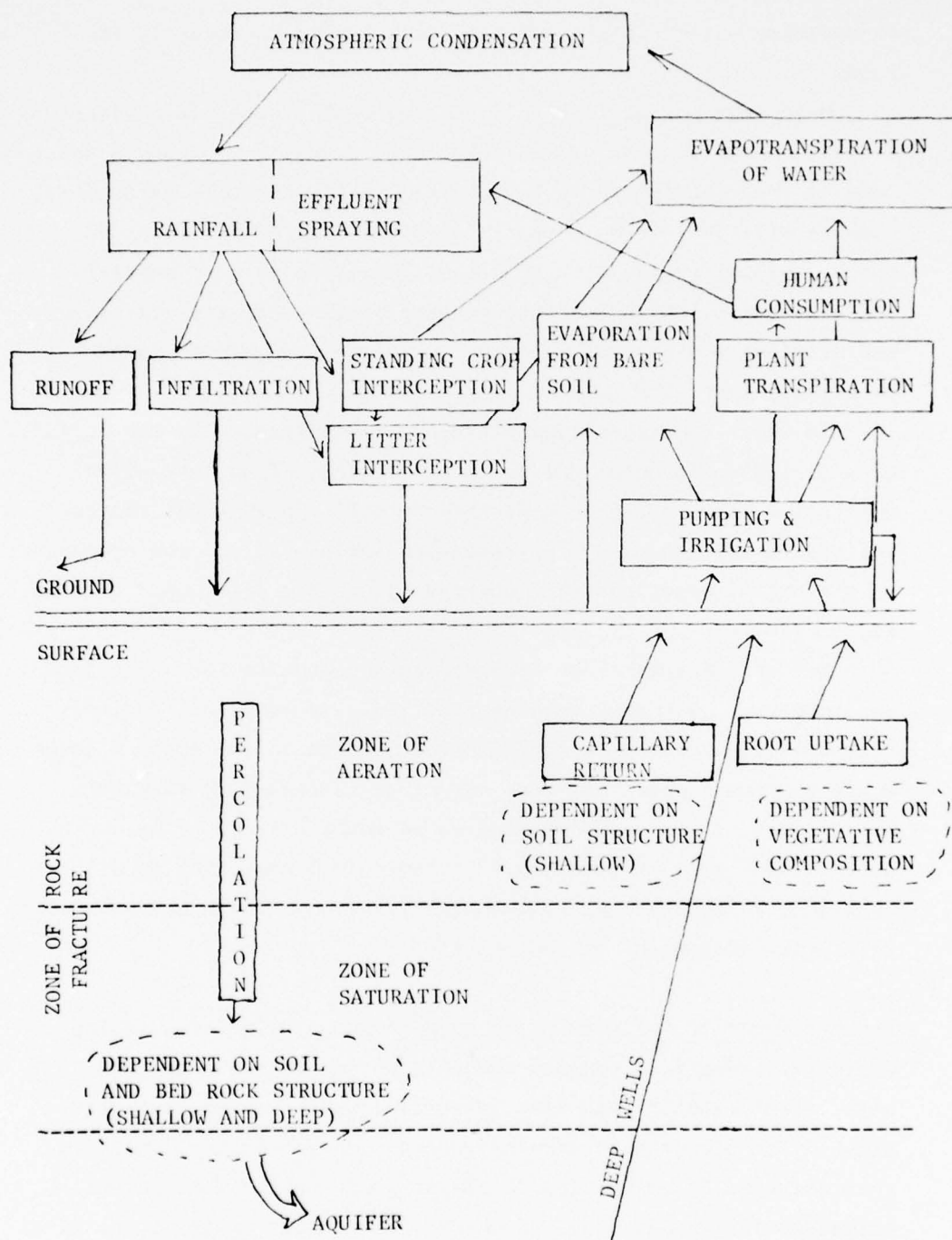


Figure 13. Conceptual representation of water pathways through a terrestrial ecosystem (modified from U.S.D.A., 1955).



recognition and definition of long range goals is securely in hand.

Major changes will occur in terrestrial systems if subjected to large scale vegetative manipulation. Conceptual pathways for some of these changes are shown in Figure 14. Drastic perturbation results in virtual curtailment of biological influence on the hydrologic cycle, with resultant increases in soil particle and nutrient deletion. At the extreme, soil sterility will result and nutrient and particle loading on adjacent waterways is unacceptably increased.

In addition, vegetative patterns are influenced by the actions of a biologically dependent group of species. Thus, any given vegetative pattern can be explained by soil and physical characteristics, climate, man's perturbation influence, and the community of biological organisms that utilize plants for life support (Figure 15).

Current biological knowledge does not provide for a detailed deterministic view of ecosystem function. It is clear, however, that our economic and technological abilities to implement a large scale waste management systems are minor obstacles to long term maintenance of man's environment as we would like it to be. Restrictions will be determined by biological realities of trophic dynamics, maintenance of system stability, and evolutionary capabilities for development and adaptation.

Trophic Dynamics and Population Regulation: The natural living world is arranged in complex channels of supply known as food webs. Phototrophic organisms (plants) convert incoming abiotic stimuli (sunlight) into chemical energy (carbohydrate), which is then consumed by successive levels of plant-eating animals and animal-eating animals, all of which subsequently revert to original states via decomposer components. Entrophy changes appear to constitute the only irreversable deletions on a geological time scale.

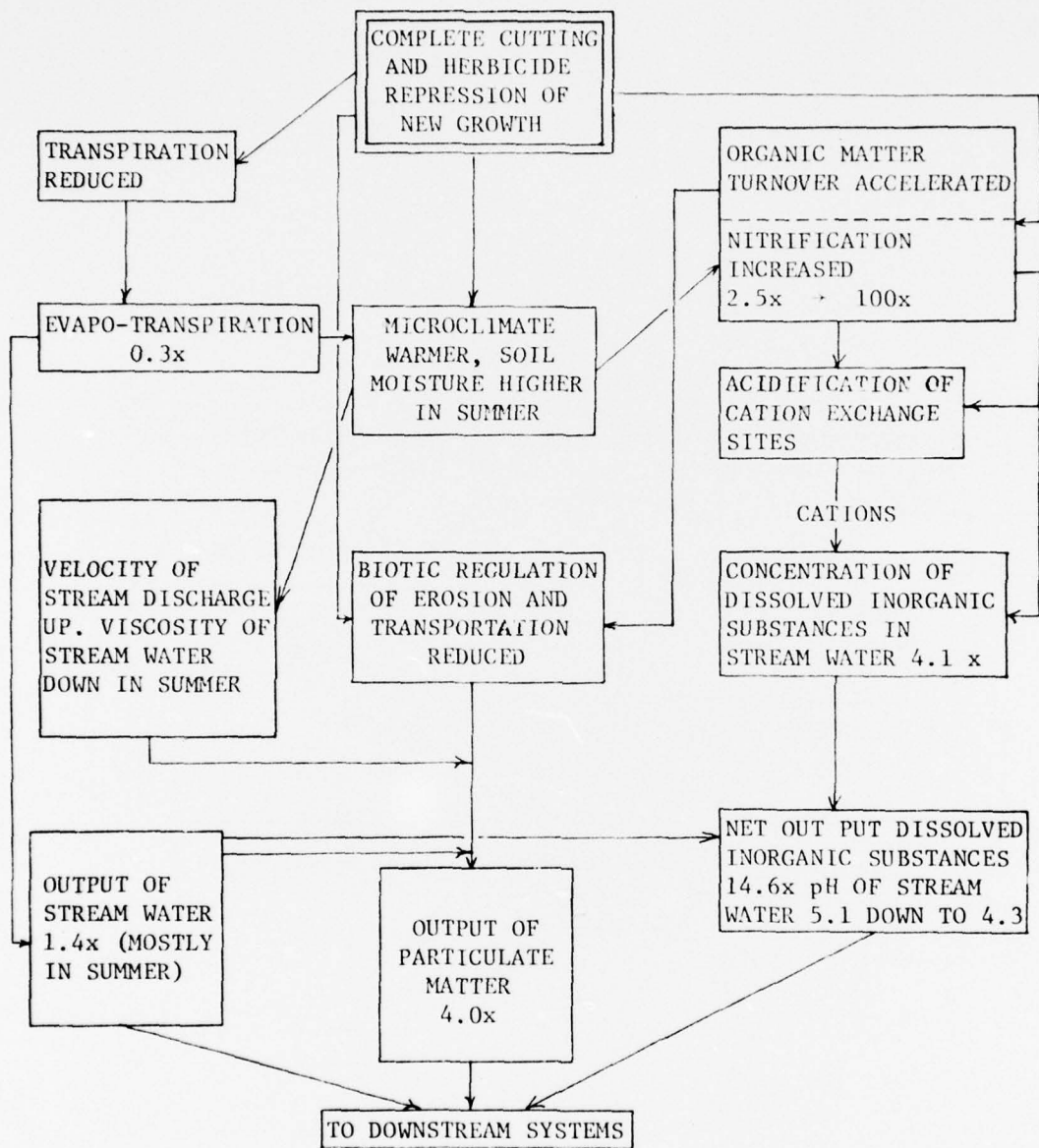


Figure 14. Ecological effects of deforestation of a watershed (from Likens and Bormann, 1972). Harvesting, plowing, and replanting of a terrestrial system subjected to effluent input will create a comparable events chain.

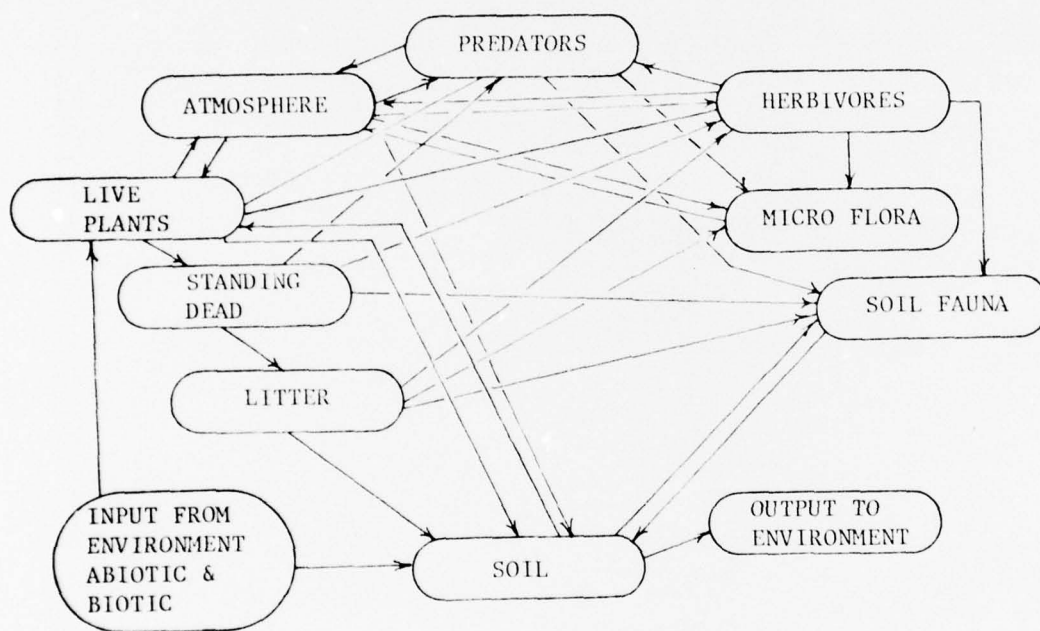


Figure 15. Homomorphic macro-model of a conceptualized terrestrial ecosystem. (Modified from Schultz, 1970). Arrows Indicate Paths of Nutrient and/or Energy Flow.

On a shorter time scale potentially hazardous non-metabolizable materials (i.e. heavy metals and ecologically foreign and refractory chemical compounds) must not be permitted to build up within the system. In addition, deposition of metabolizable materials on lake and sea beds must be avoided, for this process constitutes an effective removal of essential materials from man's sphere of influence and time consideration (e.g. phosphorus depletion).

A conceptual food web structure for a typical terrestrial ecosystem is shown in Figure 16. The foundation for such a food web are the primary producers (plants), which are dependent upon the physical environment. Each of the categories radiating from the live plant module may contain anywhere from several to perhaps a hundred individual species, each serving as a process-pathway, thus providing many alternative modes for energy concentration and utilization. It is extremely important to point out that some of these components are more essential than others. For example, the system would not be drastically altered if the large herbivores were removed. But the removal of the pollen-feeding insects (bees, etc.) would result in subsequent deletion of numerous species of angiosperms that depend upon this activity for reproduction. Many agriculturally important crops are dependent upon insect pollinators. In other cases food chains are almost entirely vertebrate with the primary consumer being grazing rodents (lemmings, voles), with several levels of predator species as dependent populations.

Maximum turnover rates and high efficiencies in a terrestrial system implies numerous pathways for processing energy. Maintenance of habitat patterns that allow this complexity cannot be overstressed. In spite of our technological ability to artificially drive man-made communities, our mastery is only superficial.

Even though primary production is ultimately one of the main factors controlling the rates of multiplication and growth of organisms in a community (Westlake, 1963), organisms with complex behavioral patterns have developed self-regulating mechanisms that preclude the effects of starvation. Territoriality is such a mechanism whereby organisms "sense" high densities, and can thus

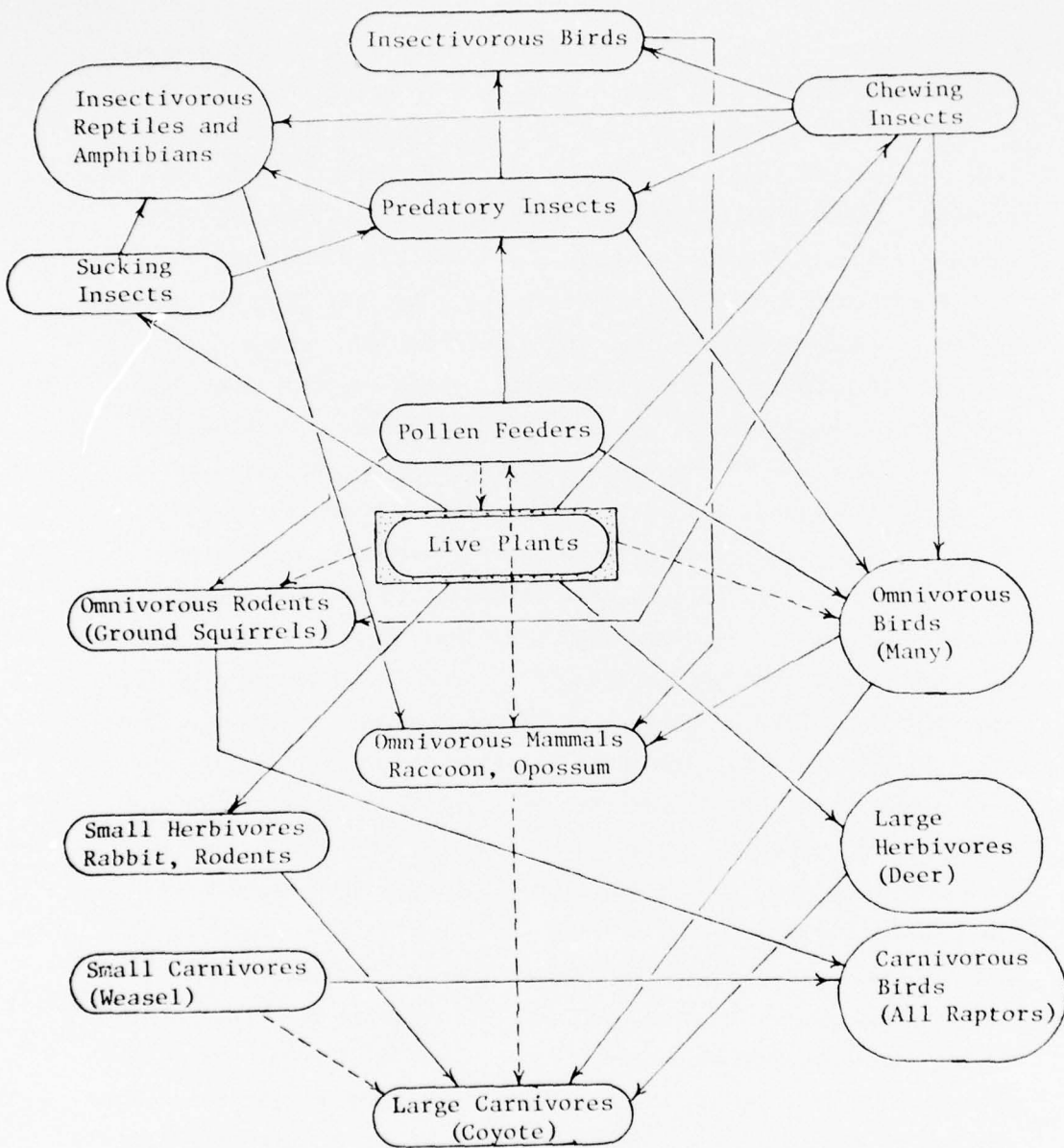


Figure 16. Conceptual food web structure for a terrestrial ecosystem. Perturbations affecting live plant level will have strong influence on overall trophic structure.



maintain at least a continuing breeding population. Most forms of environmental perturbation will interfere with these regulatory mechanisms. A deer population is one example of a species that flourishes in early successional stages, but starves and decreases when rigid maintenance programs are placed into effect. A schematic of regulatory pathways is shown in Figure 17 for a typical deer population (a similar scheme could be drawn for any vertebrate population). Regulation in such populations can be promoted through two pathways, habitat control and direct population control. Habitat control is most effective as a natality value, while direct population input is most simple via hunting pressures (thereby sex and age structure modification).

#### Diversity and Stability

Barring man's influence, ecosystems reflect the physical environment in which they have developed. From bare substrate, a system may be expected to develop through a rapid growth stage that leads to an oscillating steady state. Early successional stages are characterized by high production/respiration ratios, high yields (net production), short food chains, low diversity, small sized organisms, open nutrient cycles and a lack of stability (Odum, 1972).

The concept that regional vegetative patterns tend towards a single climax vegetative community as a direct result of climate (i.e. monoclimate hypothesis of Clements, 1936), has been largely replaced by the realization that edaphic factors, nutrients, soil moisture and physiography all play an important role in determination of a final steady state (i.e. polyclimate hypothesis of Tansley, 1939; also see Daubenmire, 1966). A management paradox arises, inasmuch as desirable attributes lie at both ends of the successional spectrum. Young communities have high production and net yield but low complexity and thereby fewer process-pathways. They have low stability (thereby reliability), but are more agreeable to short term harvest strategies. Climax stages have high diversity, complex process-pathways and high stability, but low production.

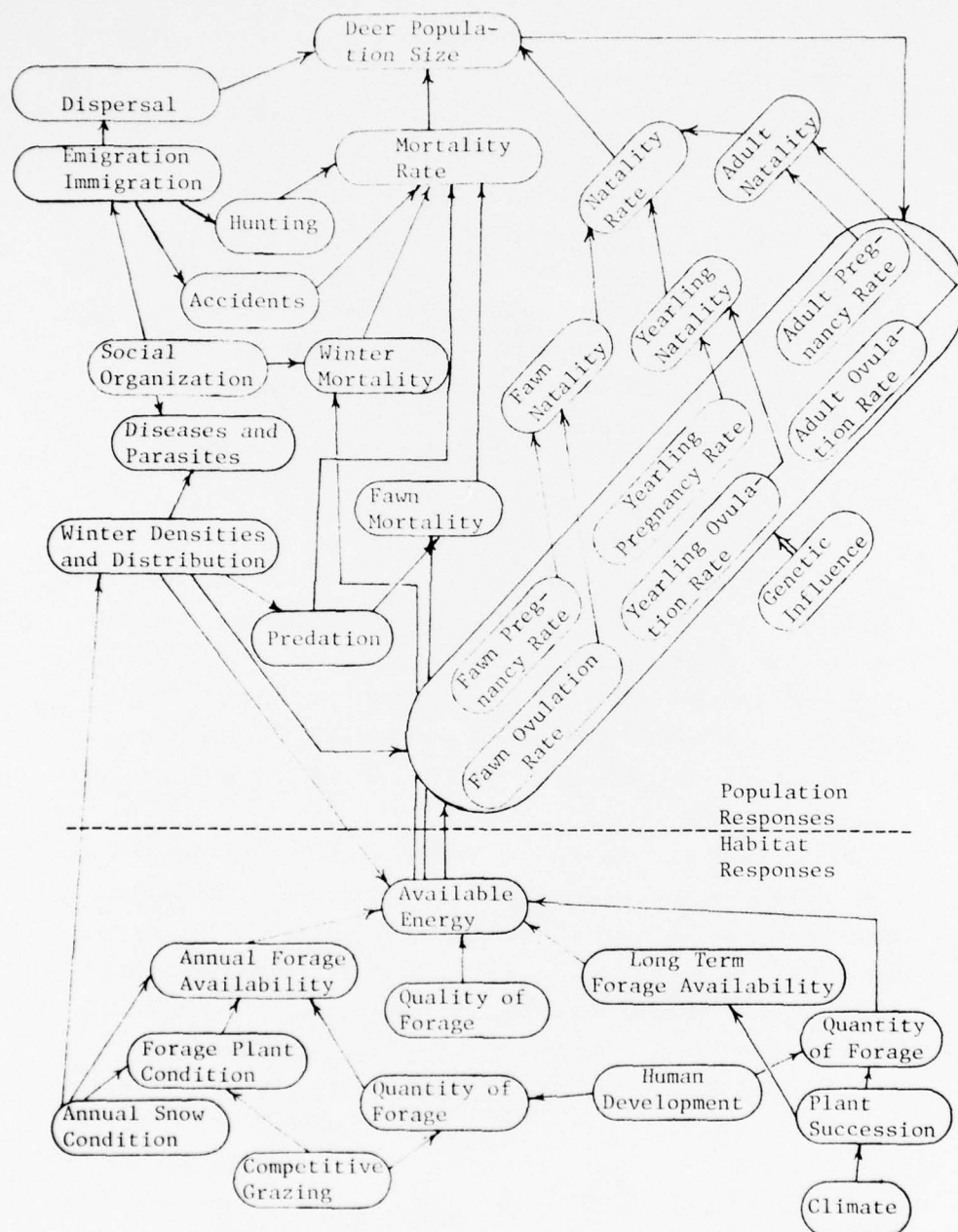


Figure 17. Schematic diagram of factors promoting regulation in a typical big-game population. Usual management procedures are involved with manipulation and control of habitat responses, and via control of mortality rate through hunting pressures.

Since high productivity with harvest is essential for proper maintenance of a wastewater irrigation system, potential problems arising from low diversity and stability must be resolved.

At least six sorts of evidence support the concept that the balance of simple communities is more easily upset than that in more complex ones (Elton, 1958).

First are conclusions based on mathematical speculation about population dynamics. Simplified models of predator-prey systems, or of two competing species conspicuously show strong oscillations. In the rare case where these systems do not terminate in extinction of one or both populations, constant densities are not obtained, but rather these populations are subject to periodic "outbreak" densities. Experimental evidence currently supports this conclusion (Gause, 1934, Park, 1948).

Second, the addition of more complex structure providing cover and habitat diversity tends to reduce density fluctuations, and large scale habitat diversity that provides a source for reinvasion provides an additional damping effect.

Third, evidence indicates that low diversity natural habitats on islands are much more vulnerable to invading species than communities on the continents. Deer and goat populations on New Zealand, and rabbits and cactus on Australia are well known examples.

Fourth, invasions and outbreaks most often happen on cultivated or planted lands. Complex, mature communities possess the ability to repulse incoming species by denying an opportunity for initial establishment.

Fifth, insect and pest outbreaks are virtually unknown from undisturbed tropical rainforests. Only when these forests are removed and simplified do pest problems become serious. The theory implies that there are always enough enemies and parasites available to turn on any rapidly increasing species. Thus, by a complex system of checks and buffers, such species are soon put back in their place.

Finally, a sixth kind of evidence comes from research in orchard pest control (Pickett, 1949; Pickett and Patterson, 1953). Spraying programs that eliminate parasite and predator species as well as the target species inevitably result in subsequent outbreak of the undesirable forms. Pest species typically possess high fecundity owing to their status as prey species for a variety of dependent species. High fecundity rates provide two advantages. When pressure is relaxed, they increase at phenomenal rates; and, short generation time and high turnover provide rapid adaptation to specific chemical compounds used for control.

Formal models have been constructed to explain and verify much of the above discussion (see MacArthur, 1955, 1969; Pielou, 1966; Preston, 1962). Contrasting to the above discussion, it should be recognized that few things in ecological systems are absolute. The world contains thousands of square miles being maintained in essential monoculture. We must inquire why it is that such systems are duplicated year after year with a relatively small amount of the total being periodically ruined by a pest outbreak. The answer, while obvious, is of great importance to man's continued ability to maintain status-quo. Simple, low-diversity, unstable communities are being sharply driven by a constant external control. Agricultural practices are designed to rejuvenate depleted soil constituents. Pest species problems are being controlled by evermore toxic compounds, and most important, there still remains sufficient peripheral diversity to provide reservoir sources for reinvasion of naturally occurring enemies of our major pest species. In some cases of introduced pest species, partial success has come through subsequent artificial encouragement of a specific predator or parasite (i.e. artificially increasing diversity and complexity).

At best, in a massive terrestrial wastewater irrigation effort, we must expect to have to drive the system rigorously. Enlightened control programs will require maximum possible ecological complexity, even if artificially maintained. Maintenance of this complexity will probably not be concurrently the most economically efficient,

nor will it necessarily provide the highest short term production and processing capabilities. The alternatives, however, are an increased instability where one must expect periodic devastation by uncontrollable pest species.

It should also be recognized that some of these pest species will be vectors for pathogens directly harmful to man's health and well-being. Mosquito populations, and rodent-flea combinations are still capable of producing epidemics even in the United States. Witness the recent outbreak of equine encephalitis in Texas, or the continuing (though low-level) threat of bubonic plague in the American southwest. Total chemical control is contrary to long term goals, and thus unacceptable. Some chemical control coupled with maintenance of a healthy, diverse ecosystem structure may be an initial approach to a solution.

The portion of a terrestrial ecosystem that would eventually receive widespread application of wastewater can be looked at from two viewpoints: first, as a discrete wastewater processing system artificially separated from the rest of the ecosystem by various physical means. Here, the land, with its assemblage of biotic and abiotic elements, is nothing more than another waste treatment plant; "dirty" wastewater enters the system, and "clean" water leaves the system. What happens within the boundaries of the system is of little concern as long as the output (clean water and recyclible biproducts) is satisfactory. This viewpoint provides an easy conceptualization upon which to base the environmental impact of land disposal. Namely, the enclosed area of the land disposal system is profoundly changed in terms of stability and diversity and is, in essence, sacrificed for a limited use. And as long as the system performs according to design standards its impact on adjacent areas will be minimal. This line of reasoning is simplistic and unrealistic in light of the dependance of system function on biological processes. We chose to envision the land disposal scheme from a second viewpoint; that of being an integral portion of the terrestrial ecosystem as a whole. In light of this choice we will consider not only the environmental



impacts of land disposal on adjacent areas but the environmental impacts with the boundaries of the sites.

For further insight on the global considerations of intensely managed ecosystems the reader is referred to "Recycling and Ecosystem Response to Water Manipulation" (Bahr, Cole and Stevens, 1972). This report establishes the inseparable ties between terrestrial and aquatic systems.

#### Waterfowl Ecology

The basic needs of any animal species can be simply identified as food, water, and shelter. The needs of the 25 waterfowl species present on the Lake St. Clair-Lake Erie areas are varied, depending on the species. Water is a critical component and a wide variety of water areas will provide the shelter function for most species. A broad range of plant and animal material serve as important waterfowl foods (Table 39). Although these represent a wide range of plant and animal communities, they are generally found in areas where the water varies in depth from 0 to 12 feet. Thus, most important waterfowl habitats occur within 0-12 feet deep zones of wetland areas. Most differences in habitat utilization between species begin to appear when food habits are considered. The needs are generally divided according to the waterfowl group and focus on food preferences. The habitat zones from a continuum ranging from the shallow water zones beginning at the interface between land and water to the open water areas 10 to 15 feet deep. Geese and swans feed almost exclusively on plants, especially on vegetative portions such as foliage and stems (Martin, et al., 1959). Members of the dabbling duck group are principally plant feeders with 80 to 90% of the diet consisting of plant materials (Martin, et al., 1951). Both of these groups depend on the water zone from 0 to 3 feet deep. Members of the diving duck group consume both plant and animal materials with 20 to 80% of the diet consisting of plant materials (Martin, et al., 1951). This group depends on the water zone from 1 to 12 feet

Table 39. Important foods of game ducks in the eastern region of the United States (data from Martin and Uhler, 1939).

Plant Material		Animal Material	
Scientific Name	Common Name	Scientific Name	Common Name
<u>Potamogeton</u>	Pondweed	<u>Insecta</u>	Insects
<u>Polygonum</u>	Smartweed	<u>Gastropoda</u>	Snails
<u>Zizania aquatica</u>	Wildrice	<u>Crustacea</u>	Crustaceans
<u>Scirpus</u>	Bulrush	<u>Pelecypoda</u>	Bivalves
<u>Najas flexilis</u>	Northern naiad	<u>Pisces</u>	Fishes
<u>Lemna, Spirodela, et al</u>	Duckweed		
<u>Vallisneria spiralis</u>	Wildcelery		
<u>Zea mays</u>	Corn		
<u>Leersia</u>	Cutgrass		
<u>Chara et al</u>	Muskgrass		
<u>Setaria</u>	Pigeongrass		
<u>Echinochola</u>	Wild millet		
<u>Fagopyrum esculentum</u>	Buckwheat		
<u>Sparganium</u>	Burreed		
<u>Carex</u>	Sedge		
<u>Sagittaria</u>	Arrowhead		

deep. The mergansers primarily consume fish or other animal materials with plant foods being negligible (Martin, et al., 1951). They generally occupy the water zone from 3 to 12 feet deep.

The most important areas in the Lake Erie-Lake St. Clair region for geese, swans, and dabbling ducks are the marsh areas along the edge of the lakes which includes the Anchor Bay area in Lake St. Clair, Point Maumellee area in the lower Detroit River, and the Monroe and Maumee Bay areas of western Lake Erie. The diving ducks depend on the open water areas from 0 to 12 feet deep. The two most important areas are the Anchor Bay region of Lake St. Clair and the lower Detroit River and adjacent areas of Lake Erie.

The lower Detroit River and adjacent waters of Lake Erie is suitable waterfowl habitat during the migratory and winter periods. The important submersed plants in this region are Wildcelery, Berchfold's pondweed, sago pondweed, water-weed, water milfoil, muskgrass, mud plantain, arrowhead, and Illinois pondweed (Hunt, 1957). There are three important aquatic invertebrate communities, tubificid worm, snail, and fingernail clam (Hunt, 1957). Wildcelery tubers and mollusks are the most important foods for black ducks, mallards, canvasback, redhead, and scaup. The standing crop of edible portions of wild celery and mollusks was estimated by Hunt (1957). He concluded that a single black duck or canvasback would find food for 4.6 million days, or a redhead for 7.6 million days, or a scaup for 21 million days. This suggests that the annual waterfowl use of this area of 15.7 million days is close to the carrying capacity of the region.

The distribution and number of waterfowl using the Lake Erie-Lake St. Clair region is influenced by a number of factors. The population numbers are checked by environmental limitations, including the presence or absence of food, disease, predation, strife and lack of space. Food appears to be the most important factor influencing the numbers and distribution of waterfowl in this region. Within areas where food is generally available, human activities associated with various recreational pursuits

and industrial activities may cause minor shifts although they do not appear to be important. Waterfowl hunting will cause the most obvious shifts in distribution (Hunt, 1957; Reed, 1971). Reed (1971) observed that the birds were located further ashore during the hunting season compared to the period just after the hunting season or the spring. Ice formation during the winter over most of the region does concentrate the dabbling and diving ducks into the lower Detroit River area. Reed (1971) observed a flock of mallards and black ducks numbering 5,000 to 7,000 birds near Erie, Michigan. The birds used the open water in Lake Erie resulting from the discharge of cooling water of a fossil fuel electric power plant and a natural spring about 1 mile away.

Data concerning waterfowl mortality for the region are qualitative at best. The best information available concerns the mortality resulting from hunting. Natural mortality in the region during the winter has been documented by Hunt (1957). He reported that mortality of ducks on the lower Detroit River varied from an estimated 100 birds to 10,000 birds. The factors causing death were starvation, 37 to 56% of the mortality; oil pollution, 9 to 33% of the mortality; and disease, 30 to 35% of the losses. These factors acted singly or in combination to cause death. Included under the category of disease was lead poisoning, aspergillosis, coccidiosis, helminthic infestation, phosphorus poisoning, pneumonia, mechanical injury, proventriculitis, and tumor. Oils, externally applied or ingested, result in harmful effects on ducks, especially during periods of cold weather. The oil reduces the insulating quality of the plumage and the metabolic rate increases to produce more heat resulting in accelerated starvation (Hunt, 1957; Hartung, 1965, 1967; Hartung and Hunt, 1966). The major food beds in the lower Detroit River become covered with ice some years. Hunt (1957) notes that death from starvation begins 1 to 2 weeks after 70 to 90% of the food beds are covered. Data concerning natural mortality during other periods of the year are not available. Deaths from botulism occurs some years during the late summer in the marsh zone.

Hunting is a major mortality factor during the fall (Table 40). During the past 11 years an average of 12,500 waterfowl hunters harvested 75,300 ducks in this regions. This amounts to 17.8% of the waterfowl hunters in Michigan accounting for 18.6% of the Total waterfowl harvest. There has been a trend for the number of hunters and kill to increase during the past 11 years. The most important areas are Anchor Bay region of Lake St. Clair, lower Detroit River, and the shoreline of western Lake Erie from Detroit to Maumee Bay. Waterfowl hunting should be considered an important recreation activity during the fall.

#### Stream Ecology

Streams are simply nature's system of carrying excess runoff water back to the oceans, thus serving as the physical surface connections for the hydrological cycle. Water is collected and delivered both from direct runoff and indirectly from ground water and soil sources where such water issues forth upon the earth's surface.

Obeying the law of gravity, the water moves unrelentlessly downhill and carves the channels of the streams in the process. Because of the differential resistances of materials encountered, the water erodes the channels unevenly and in a meandering fashion. This results in a variety of habitats such as pools, riffles and runs. It tends to maximize the stream bottom area and results in increasing the living space for organisms.

The stream serves as a carrier for materials that are water-soluble or that are light enough to be swept along by the current. This action provides nutrients for the attached plants of the stream and brings food to the animal populations. Because of the unidirectional flow of water in streams there exists an open and relatively non-cyclic system with regard to nutrients. Nutrients unused or released at one point continue to move downstream. Streams thus have less opportunity to store nutrients than do lakes. Plant life in streams can survive on lower concentrations of nutrients for two reasons, the supply is continuous from upstream and



Table 40. The estimated number of waterfowl hunters and ducks bagged in Monroe, Wayne, Macomb, and St. Clair counties of Michigan from 1961 through 1971 (data from Michigan Department of Natural Resources, Research and Development Division).

Year	Number of Waterfowl Hunters	Percentage of Total Waterfowl Hunters in Michigan	Number of Ducks Bagged	Percentage of Total Harvest in Michigan
1961	8842	17.4	50700	19.5
1962	6818	17.5	27700	17.5
1963	8940	16.7	28600	11.5
1964	10923	17.1	69300	18.9
1965	11708	18.5	80600	25.3
1966	15812	22.3	108400	23.4
1967	16327	20.8	96700	19.8
1968	12381	16.5	43200	14.0
1969	13348	14.7	100400	19.9
1970	17243	15.3	150600	22.7
1971	15081	18.8	72300	12.3
Mean	12493	17.8	75300	18.6

the water current reduces the concentration gradient between the living cell surface and the nutrient molecule. Thus we often see algal mats or weed beds existing in streams on lower nutrient concentrations than would be required in standing waters.

The life in streams as elsewhere is governed by the principle of Limiting Factors. As long as the basic nutritional requirements are met (Liebig's Law of the Minimum) and the tolerances of the organism are not exceeded (Shelford's Law of Tolerance) the organism can survive. These two laws are of primary concern in this report and will be used as the basis for judgements regarding the presence or absence of species. The size of populations and complexities of communities depend upon the quantity of energy available once the environmental quality is satisfactory, thus population numbers will be of secondary importance in this report.

In streams the most influential parameters seem to be sediment load, temperature, dissolved oxygen and toxic concentrations of materials. Nutrient and non-living organic materials are important as they influence the dissolved oxygen concentration.

While sediments are physically important as they cover spawning beds and bottom dwelling invertebrates, they are of minor concern here where we are focusing on the effectiveness of pollution control methods aimed largely at non-sediment aspects.

Temperature is an important regulator and will be discussed as it is involved indirectly in alternative methods of wastewater treatment. The alternative treatment method of spraying wastewater on the land may tend to moderate stream temperatures if drainage water is diverted to streams. Here, one would expect cooler water temperatures in summer and warmer water temperatures in winter. This would favor game-fish species.

Toxic concentrations of such things as heavy metals and pesticides are important in limiting the presence or absence of species. The tolerance of certain fish species may be exceeded and thus those species are eliminated directly or certain food organisms, perhaps insects, may be killed and thus the fish are eliminated indirectly because of an insufficient energy base. Toxic relationships were

discussed earlier.

Perhaps the most important parameter is the concentration of dissolved oxygen and those factors that influence that concentration. Many aquatic organisms have been classified as tolerant of pollution and this largely on the basis of their oxygen requirements. If the stream fails to supply the minimum requirements of the organism for oxygen, it will perish. The species will not survive if any of its life history stages fail and, of course, oxygen must be present continuously. Even the lack of oxygen for an hour or so at anytime of the day can be lethal.

Thus, dissolved oxygen sags which occur at night during the summer can be as effective in limiting certain species as if the oxygen were gone throughout the year. Diurnal oxygen sags result from plant growth caused by excessive nutrient supply. While the plant mass provides oxygen by photosynthesis during the day, the majority of the oxygen diffuses out of the water as it exceeds saturation. Respiration (without photosynthesis) at night consumes oxygen often reducing the concentration below fish requirements. Likewise the disposal of organic matter into streams provides a food base for bacteria which also consume oxygen during their respiration (BOD).

#### Oxygen and Temperature Requirements of Aquatic Organisms

It is reasonable to discuss the oxygen and temperature requirements of aquatic organisms together since the two parameters have a considerable influence on one another, especially temperature on oxygen. As water temperature increases the saturation concentration is reduced, thus it would appear that warm water holds less oxygen than cold water. The literature is full of arguments supporting this view and the contrary point that it is not the concentration of oxygen that is important but rather the partial pressure. In other words, a given percent of saturation in warm water is just as available as that same percent of saturation in colder water even though the warm water concentration is less. This is further compounded by the increase in the metabolism of

cold-blooded species as the water temperature increases, thus causing a greater demand for oxygen. The end result of these complexities has been a failure to establish exact oxygen requirements for most aquatic animals. There are, however, general requirements that have been established and accepted as such by most ecologists.

As an example, the upper temperature level for adult brown trout is about 23 C with brook trout and rainbow trout usually tolerating slightly lower temperatures (Harlan and Speaker, 1956). Smallmouth bass will survive environmental temperatures of 33 C but prefer 28 C or less (Bennett, 1965). Largemouth bass can usually tolerate temperatures slightly higher than the smallmouth. Most of the other sunfishes are in the same situation as the large and smallmouth bass. The critical factor controlling stream-dwelling smallmouth bass is more often sediments than either temperature or oxygen. It is likely that walleye spawning success is also sensitive to sediment deposition in stream situations.

The ranges of oxygen concentrations for survival for summer and winter for several species of fish has been given by Moore (1942) as observed under natural field conditions. These data are given below in Table 41.

Table 41. Lowest observed oxygen concentration (mg/liter) at which fish survived. (Moore, 1942).

Species	24-Hour Survival Summer	48-Hour Survival Winter
Pike	6.0	3.1
Black Bass	5.5	4.7
Yellow Perch	4.2	4.7
Black Bullhead	3.3	1.1

Laboratory tests have shown survival at concentrations lower than those in Table 41; however, there have been no considerations given

to overall health or stress of the fish. Rainbow trout have been shown to survive for 84 hours at oxygen concentrations as low as 2.6 at 20 C (Water Pollution Research Board of England, 1956), but simple survival is not the same as maintenance of a healthy condition. Ellis as early as 1937 stated that dissolved oxygen concentrations should be at 5.0 mg/liter or higher to maintain a healthy and varied fish fauna. It is probable that trout need 6.0 mg/liter for "good" conditions.

Data for the oxygen requirements of invertebrate aquatic organisms are less available. Many ecologists have categorized these organisms as tolerant, intolerant, and facultative. Tolerant is defined as growing and developing within a wide range of environmental conditions and intolerant within a narrow range. Intolerant are rarely found in areas of organic enrichment and are especially sensitive to low oxygen concentrations. Facultative organisms are intermediate, tolerating all but the severe conditions.

Good data are available for the caddisflies which can be considered generally as intolerant. Using the caddisflies as typical of the intolerant, the data can be extended broadly to other groups. Roback (1962) reported for some 20 species of caddisflies that they occurred naturally far more often at or above 7 mgO<sub>2</sub>/liter than below. The frequency of occurrence being 13 below 7 mg/liter compared to nearly 600 at or above 7 mg/liter.

#### Uptake of Heavy Metals Into Plants

Earlier in this report we discussed the presence of heavy metals in aquatic systems. Here we would like to explain an important process in the translocation of these potentially hazardous materials into higher levels of the food chain; the uptake of these materials into the base of the food chain.

Plants will accumulate each of the elements that are present in their environment. Bowen (1966 p. 88) shows concentration factors on the order of  $2 - 4 \times 10^3$  for Hg, Fe, Zn, Co and Cd and in that order of affinity. He cites the mean values of Russian workers who studied the uptake of 19 elements from radioactively labelled



freshwater by 32 species of algae, bryophytes and vascular hydrophytes. The compilation of Chapman, Fisher and Pratt (1968) indicates a similar range of concentration for these metals, and others including Mn, Ni and Pb. Boyd (1968) found that plants concentrate elements from the environment primarily in the early stages of growth, trailing off in uptake as the growing season progressed. The work of Tierney (1972) shows that this is not a consistent pattern for heavy metals as a group.

Information in the literature indicates that, within the limits of toxicity, a metal will be absorbed in an increasing amount as its abundance is increased in the environment. Dolar, Keeney and Chesters (1971) have demonstrated this with a freshwater vascular plant; Knauss and Porter (1954) found that the log of the concentration in Chlorella cells was proportional to the log of the concentration in the medium over a hundred fold range in concentration for Ca, Cu, Fe, Mn, Sr and Zn; Chiaudoni (1969) observed a similar relationship for Cu in stems and roots of an emergent marsh species; Hutchinson (1972) and Rolfe, Melin and Ewing (1972) have observed this pattern for Pb in terrestrial systems. The data of Table 42 are consistent with this view. As with a large segment of the current literature, the concentrations of elements in the environment was not available from this work. However, the pattern in the data is consistent with observations of the composition of natural drainage to Lake Karapiro and the nature of the effluents entering the two systems. The Auckland (New Zealand) ponds are particularly enriched with Cr, Cu and Ni originating in sewered metal plating industries in the city. Notice from the table that this increase can be one order of magnitude. The difference in these two environments likely represents a maximum covering the range from near oligotrophy (Lake Karapiro) to hypereutrophy (Auckland ponds). Differences observable in natural populations of a species of aquatic plant are not likely to be greater than this.

Understanding of the mechanisms controlling the percentage of the metal that is available from the total supply in the environment

Table 42. Changes in the tissue concentrations of heavy metals in shoots of Egeria densa transplanted from Lake Karapiro, New Zealand, and grown in waste stabilization ponds at Auckland for 28 days (from McNabb and Tierney, 1972).

Element	Initial Concentration µg atoms/Kg	Final Concentration µg atoms/Kg
Cr	154	1077
Ni	136	903
Co	289	594
Cu	189	2598
Zn	229	428
Cd	44	62
Pb	241	627

is in need of rigorous modeling founded on a combination of theoretical expectation and experimental data. This is particularly true for the aquatic system and for all highly polluted systems in which the variety and quantity of chelators may significantly shift the availability of metals. For this reason, it is impractical to attempt to assess the potential effect on uptake of element interactions. It can be said that competition occurs between pairs of similar ions, for example, sulphate and selenate or phosphate and arsenate (Fried and Shapira, 1961). In these cases, excess of one ion in the medium depresses the uptake of the other. Uptake can be disorganized by unsuitable ratios of cations as well (cf. Bowen, 1966 p. 94; Treshow, 1970 p. 176).

The uptake of metals in plants is influenced by the relationship of structures in the plants to the medium in which the metals occur. Planktonic plants (algae) are frequently enveloped by organic polymeric excretions that can function as ion exchangers. Beneath this lies the cell wall in which sequestration may occur as cited earlier. Movement of metals across the interior cell membrane appears to be accomplished primarily by active transport. This process occurs in cells that are freely respiring an energy source in the presence of oxygen. Bowen (1966 pp. 91-93) provides a brief discussion of this. As a result of their suspended state when actively metabolizing, planktonic algae and free floating vascular plants (as Lemna) have access to elements in the water, but not to those in the sediments.

Benthic algae (as Cladophora) or rooted vascular hydrophytes that are submersed in water may absorb elements from either the substrate or the ambient water. Table 43 shows that the concentration of heavy metals in these two phases is quite different; the sediments are 3 to 4 orders of magnitude higher in both the urbanized and the non-urbanized environment. Because of this, the question of whether these plants absorb metals primarily through their roots and rhizoids or through their foliage is of importance. Studies that have been done to date have been concerned mostly with phosphorus rather than heavy metals. Bristow and Whitcombe (1970)

Table 43. Heavy metals ( $\mu\text{g atoms/Kg}$ ) in water and sediments of an urbanized and a non-urbanized stream in central Illinois (from Mathis and Cummings, 1971).

Element	Urbanized Illinois River		Non-Urbanized Lamarsh Creek	
	Water	Sediment	Water	Sediment
Cr	0.38	596	0.09	104
Ni	0.05	767	0.02	262
Co	0.05	136	0.01	109
Cu	0.02	441		99
Zn	0.76	1942	0.35	443
Cd	0.004	37	0.001	3
Pb	0.010	188	0.002	71

present data to indicate that root uptake is more important than foliar uptake of P in species of Myriophyllum and Elodea. Waisel and Shapira (1971), using the same species of Myriophyllum and two species of Potamogeton, have shown that shoots are self-sustaining and content that roots do not play a critical role in absorption and translocation of ions. Coffey (1972) has demonstrated that dependence on root absorption varies among species. It was found that in working with transplanted plots having the same soil but placed in environments differing widely in ambient water concentrations of P (McNabb and Tierney, 1972), that the submerged plant is dependent on the hydrosol for nutrients only in oligotrophic environments. In these, correlation does not exist between tissue concentration and ambient water concentration (cf. Livingston and Bentley, 1970). Relatively slight enrichment of the ambient water shifts the dependence such that this and tissue concentration are strongly correlated in a positive manner. McNabb is presently studying this relationship in regard to heavy metals; there is no reason now to suspect a pattern different from that observed for P. Foliar uptake is likely to be the important pathway for the movement of heavy metals into submerged vascular hydrophytes in waste enriched environments. The movement of currents past foliar tissue effectively raises the amount of an element available to the plant beyond that indicated by the concentration analyzed in grab samples of water. This may be among the reasons for the observation of Tan (1970) and others (cf. Tierney, 1972) that minerals determined in submersed aquatic plants are higher in concentration by weight than in grasses and other forage crops.

Caines (1965) is among those who have shown that emergent aquatic plants are more tightly coupled to soil absorption of elements than to ambient water absorption. This of course is the case for terrestrial vegetation. Antonovics, et al (1971) provide an extensive literature review and discussion of uptake in plants of terrestrial ecosystems. An important difference in the manner in which oxygen, for active uptake, reaches the roots of these two groups does exist; air canals are present in emergent (as well as



in submersed) vascular plants that transport oxygen to the root tissues. In this manner they may respire aerobically in hydrosols that are frequently anaerobic. It is important to note that foliar absorption of heavy metals from ladden water that is spray-dispersed in waste irrigation systems may change the otherwise expected concentrations in emergent or terrestrial plants thus treated. This is discussed in the land disposal phase of our report.

The plant itself is a factor that influences the uptake of heavy metals. Figure 18 is a stylized diagram to illustrate this. The relationship between levels in the shoot at increasing levels in the soil differs among the elements. The initial gradual slope for Cu, Ni and Pb as compared to Zn suggests that the former are either in less available form in the soil, are excluded by root membranes, or are stored in the roots and are not translocated to the shoots. The sharp rise to lethal levels for these elements is interpreted as a swamping of trapping mechanisms in non-living parts of the plant; in particular, cell walls. The figure suggests that plants differ in their affinity for various metals.

#### Limiting Nutrients in Lake St. Clair

The Lake St. Clair flora is a mix of species adapted to the seasonal cycles of the Temperature Zone. Growth of macrophyte and most algae populations occurs in the period April 1 to November 1 when water temperature is in the range of 10° - 25° C. While metabolic rates of some species of planktonic diatoms and flagellated Chlorophytans (green algae) can be high enough in the cold period to produce an occasional bloom, production in this interval is low.

Planktonic algae will cycle between a resting phase in the sediments and a growth phase in the overlying water. Most benthic algae and macrophytes maintain a permanent attachment to the substrate. Notable exceptions in the Lake St. Clair system are the algae Cladophora and the vascular hydrophytes Ceratophyllum demersum and Potamogeton berchtoldii (cf. Hunt, 1957). These tend to float

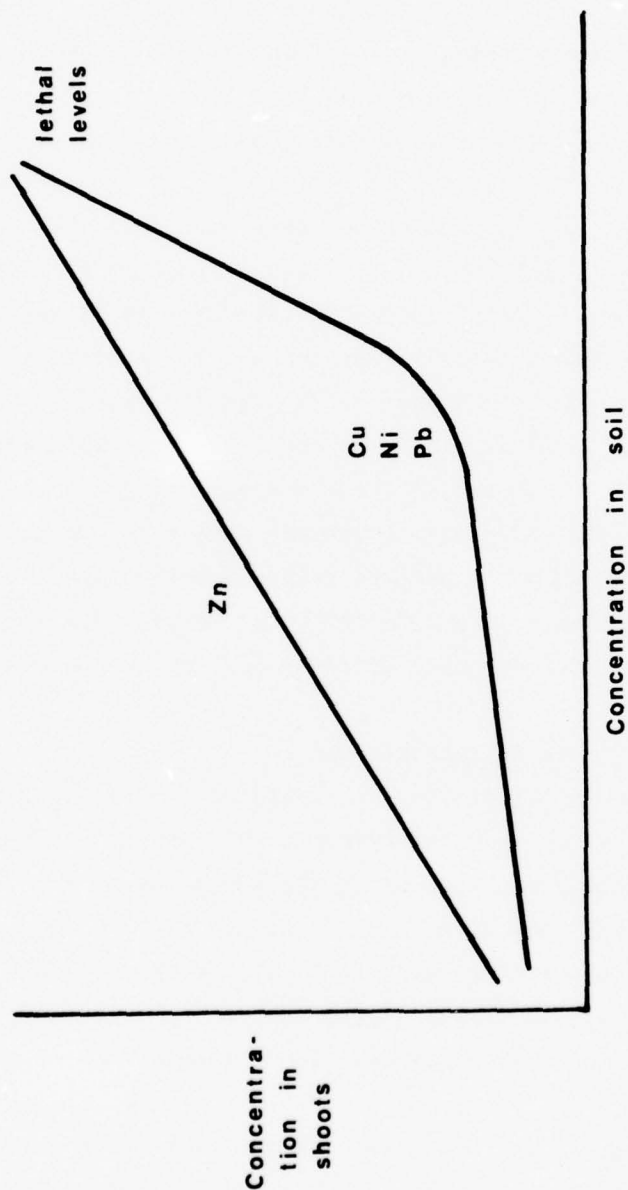


Figure 18. Generalized response of natural vegetation to increasing amounts of heavy metals in the soil (after Antonovics, et al., 1971).

to the surface in the late stages of growth and move through the lake with currents. The heavy metals they contain would be transported about in this way.

Since planktonic algae and attached plants must persist as resting structures in the sediments and begin their seasonal growth cycle there, the nature of this zone of the environment is of prime importance to their continuing success. Macrophytes are productive only in environments that are essentially aerobic during the growing season. Their roots may penetrate anaerobic zones in the sediments, but internal canals are present and oxygen concentrations tend to be maintained in the root tissues. Vegetative foliage fragments, die and decompose under prolonged anaerobiosis. Over-wintering structures are able to tolerate anaerobic conditions at low metabolic rates. Resting stages of phytoplankters are similarly able to persist in this type of sediment. Upon germination, they rise and grow in surface waters that are oxygenated during most of the diurnal cycle. Severe and continual oxygen depletion in this layer is associated with detrital concentrations that depress growth through an effect on light penetration as well. Growing phytoplankters, as well as macrophytes, are essentially aerobes. Portions of the Lake St. Clair basin that do not meet this requirement may serve as channels for dispersal of aquatic plants moving through in the currents, but will not be productive of resident populations.

Certain aspects of the light requirement for phytoplankters and macrophytes are basically different. Light is not essential to the development of bouyancy that brings germinating phytoplankters from the sediments into the well lighted surface layers (cf. McNabb, 1957). Food reserves of resting structures are metabolized to accomplish this. Such reserves are similarly used for the development of the first leaves in germinating macrophytes that are attached to the bottom. However, chlorophyll synthesis as well as continued production of carbohydrate from photosynthesis are light dependent. The depth to which macrophytes may grow in a basin is therefore limited by the availability of light energy.

Turbidity in Lake St. Clair, including that conferred by the presence of phytoplankton, may be a limiting factor for the growth of macrophytes. Since currents cycle phytoplankton in and out of the surface light zone, turbidity due to non-living suspended material (tripton or detritus) does not preclude the development of blooms.

The potential impact of toxic concentrations of metals and other nutrients on plant populations was just discussed. This is likely to occur in the Lake St. Clair system in regions strongly influenced by the inflow of waste-laden streams. However, if elements are present at toxic levels, the detrital load of the stream will generally be such that plant development will be restricted by low oxygen and the poor light relationships just discussed. As the detrital load is dropped and these parameters improve sufficiently for plant growth, metals (as well as organic materials) that may be present in toxic concentrations will tend to settle and be sequestered as inorganic precipitates or organic complexes in the sediments. Whether leaching from these sources will add a sufficient quantity of potential toxicants to that soluble or suspended fraction moving through the delta area to regions with suitable oxygen and light regimes is presently speculative.

From this discussion, it follows that heavy metals will exist in highest concentration in the Lake St. Clair basin at locations where the phytoplankton, rather than macrophytes, will be able to absorb them; namely, near the interface of stream flow and more lentic water where toxicity has declined and oxygen and turbidity relationships are tolerable. Past studies (Hunt, 1957) have shown that macrophytes are prominent only in areas away from the west shore and out of the turbid plumes of local rivers.

The water of the Lake St. Clair basin is transient, continually carrying materials (including those tied up in the phytoplankton and floating benthic plants) into Lake Erie. The sediments of Lake St. Clair serve as a trap for elements, particularly in regions not subject to scouring by the currents. The fraction

of a heavy metal likely to be tied up at the peak of the growing season in resident macrophyte populations is likely to be very small relative to the quantity trapped in the hydrosol over the interval of the growing season.

To be sure, a crop of macrophytes can accumulate quantities of metals that are potentially dangerous in the food chain. Ducks are the principle grazers in Lake St. Clair and their impact in this system was discussed earlier.

The principle fraction of macrophyte tissue will be recycled through decomposer populations following death at season's end. Riley (1939) in one of the thorough studies on metal cycles in lakes, found that the Cu he was studying dramatically pulsed in the suspended fraction of his water samples as this decomposition occurred. From the comments of Keeney (cf. Anon., 1972 ) a similar seasonal mobilization of Hg might be expected. The principal impact of the macrophyte biota of Lake St. Clair on the season cycles of metals may be similar to this. If so, a selection for the most tolerant detrital feeders among the zooplankters that are active in that season may occur in the pattern suggested by Hutchinson (1957) regarding copper.

#### Limiting Nutrients in Lake Erie

Indirect evidence based on organic material in the sediment indicates that primary productivity has increased in Lake Erie, and direct evidence indicates that standing crops of plants have also increased. A number of authors have commented on the probability that phosphorus has been the key element in the enrichment process (IJC, 1969; EPA, 1968, Anon., 1970 ). As has already been pointed out carbon, nitrogen and phosphorus have all increased although the amounts of increase are in question. Because of the relatively high carbonate alkalinity concentrations in the Great Lakes (20-25 mg/liter inorganic carbon) the proportion of increase in total carbon has been a fractional increase while the abundance of total nitrogen and phosphorus have more than doubled. Virtually no information is available for micronutrients.



Members of the Canadian Centre of Inland Waters (Anon., 1970 ) have argued, on the basis of nutrient concentrations in Lake Erie water and the nutrient concentrations found in fresh water algae, that phosphorus is the most likely limiting nutrient. They also report work conducted in the eastern basin which seems to indicate that phosphorus limitations existed in Lake Erie water at the time of that study in 1968. The study was far from conclusive however, because of "short cut" techniques and time-span limitations.

In the western basin, phosphorus concentrations are considerably greater on the average than in the central or eastern basins (EPA, 1968) or than water derived from Lake Huron (Beeton and Chandler, 1963). Mid-western basin water averages 60 µg/liter total phosphorus while mid-central basin averages 20 to 25 µg/liter of phosphorus. In lake Huron total phosphorus concentrations are about 10-15 µg/liter. Of course there is considerable variation in phosphorus concentrations throughout the lake, especially near shore but these values represent about the best estimates of average concentrations in the lake. Similarly, total nitrogen averages about 1.1 mg/liter in the western basin and 0.6 mg/liter in the central basin (EPA, 1968). There appears to be a greater recovery from phosphorus than nitrogen. Much less is known about carbon concentrations except that it is by far dominated by carbonate and bicarbonate carbon which remains fairly constant. Carbon dioxide generated by decomposition of wastes from municipalities is bound to be more important in the western basin than elsewhere because it receives 90% of the BOD that enters the lake but compared to the inorganic carbon present this is a minor difference.

Mean ratios of total carbon, nitrogen and phosphorus amount to 420:18:1 in the western basin to 1000:25:1 in the central basin while in algae the ratios run about 60:7:1. These data certainly suggest an abundance of nitrogen and carbon in the water compared to phosphorus. However, the ratios in the water of the western basin may vary in ratio from about 400:20:1 during spring when

nitrate concentrations are particularly low (Cole, 1972). In that section of the lake the total phosphorus concentrations have not varied seasonally although 100% variations are common in two-week sampling intervals. Also in the area, algal concentrations of carbon from May to October averaged between 0.5 and 1.0 mg/C/liter except during an August bloom when carbon reached 5 to 6 mg/liter (Marcus, 1972). During average summer conditions, about 20% of the available phosphorus and nitrogen was tied up in algae. During the bloom, total nitrogen and phosphorus did not change much, thus influx was thought about constant. No atmospheric nitrogen fixation took place. Apparently, the peak of the bloom was limited by phosphorus availability, but it was assumed that nitrogen fixation would have occurred if surplus phosphorus were available.

Other evidence also indicates that phosphorus is not limiting under most situations in the western basin. Gross primary productivity measured by Marcus (1972) was regularly greater ( $p < .05$ ) in a protected dredged cul de sac along the west shore than in the lake proper, even though mean total phosphorus concentrations were about the same in both locations.

Just what is limiting in the western basin is difficult to assess. Blue-green blooms are typically produced during calm warm weather in late summer on Lake Erie (Casper, 1965, Tiffany, 1958) and their frequency may be increasing. Calm periods are frequently associated with settling of solids, increased light penetration and possibly increased concentrations of carbon dioxide.

In the study area of Cole, 1970 the limiting amount of phosphorus appeared to be about 20  $\mu\text{g/liter}$ . Removal of phosphorus inputs in the western basin may require at least 60 to 70% removal to obtain a mean of 20  $\mu\text{g/liter}$ . This would have a large effect on average phytoplankton concentrations. However, any removal of phosphorus is likely to reduce the intensity of blue-green blooms if not the frequency of the blooms. Phosphorus limitation in the eastern basin is also very possible if the mean phosphorus concentration is  $< 20 \mu\text{g/liter}$ .

## SECTION V - SUMMARY OF BASELINE CONDITIONS

### Lake Erie

The input of solids into Lake Erie has increased because of human activity in the watershed and more intense changes of suspended solids have occurred in the western basin than in the central or eastern basin. Although dissolved solids may have increased by some smaller multiple than suspended solids, the extent of change is almost equal in all basins. But this only indicates the minor importance of dissolved solids in determining water quality because quality apparently improves from west to east. Quality of course is a matter of judgement; in this case it is a reflection of the "thickness" of the water - the concentration of both inorganic and organic matter in the water. But among dissolved solids, concentrations of biostimulatory elements (phosphorus and nitrogen compounds) have increased markedly along with various synthetic organics and other industrial wastes. Changes in nutrient concentrations are largely due to the accelerated use of detergent and fertilizers in the watershed. The amounts of "natural" organic wastes and coliform bacteria have increased more slowly.

The abundance of macrophyte has decreased markedly in the open bays and shallows of western Lake Erie while phytoplankton and zooplankton abundances have increased. Although there is no evidence that the abundance of zoobenthos and fish has changed their relative abundance has. Zoobenthic changes have occurred at a faster rate than indicated solely by changes in the rates of introduction of oxygen demanding organic material. Changes in relative fish abundance have been incurred due to a poorly regulated commercial fishery, increased concentrations of suspended solids, and man-caused impasses on spawning tributaries. No conclusive effects of oxygen depletion or toxic wastes have been demonstrated on fish although extensive depletion of oxygen occurs in the hypolimnion whenever it forms in the western and central

basins. Carbon content in sediments is not particularly great and has diminished in recent history in the western basin (presumably because of dilution by inorganic sediments). Therefore, it is unlikely that any increased rates of oxygen depletion that may have occurred in the recent past is due to sediment demands. It is possible that decompositional demands in the water itself have increased and extended oxygen depletion. Indirect deductions suggest that oxygen depletions may be more extensive than in the past and the cause is primarily from organic synthesis stimulated by increased nutrient availability. In the western basin no nutrient (with the possible exception of carbon as carbon dioxide) is limiting while in the eastern basin phosphorus is probably limiting.

#### Tributary Streams

The ecology of streams is based on the physical features of unidirectional flow and the ability of water to solubilize nutrients and gases. The principle of Limiting Factors controls the life in streams. Organisms survive and do well provided their tolerances are not exceeded (Shelford's Law of Tolerance) and their needs are met (Liebig's Law of the Minimum). Effects of materials or physical changes (e.g. heat) on the organisms in the streams can be interpreted in light of these laws.

Temperature and dissolved oxygen are primary factors involved in the laws. Nutrients and non-living organic matter are agents that strongly influence oxygen levels. Thus changes that affect these parameters are of prime importance. Problems arise if temperatures exceed the tolerances of the fish or supporting life and likewise if oxygen levels fall below concentrations necessary to sustain life. The high temperatures or low oxygen need not occur over long periods to be effective, but can damage populations if present for only a matter of a few hours.

The streams of southeastern Michigan (the Black, Belle, Clinton, Detroit, Huron, Pine, Raisin, Rouge and St. Claire) and the

Maumee of Ohio are typical midwestern, warm-water streams. Most show the stresses of municipal industrial and agricultural pollutants as manifest through the above laws. Based on physical, chemical and biological data, the streams are ranked as follows: the St. Clair, Black, Pine, Belle and North Branch of the Clinton are relatively good; the Huron and Main Branch of the Clinton are fair; the Rouge, Maumee and Detroit are poor; and the Raisin the worst of the group.

The streams contain varied populations of game fish such as smallmouth bass and northern pike. The headwaters of the better streams have small populations of trout. With an overall increase in water quality, the populations of intolerant invertebrates and game fish would expand considerably. Increased populations of trout could be maintained by stocking in the headwaters of the northern group of the streams. The middle and lower reaches would be good smallmouth bass streams.

Alternative treatment methods must be concerned with the flow of water in the streams under consideration as well as the quality of the water. Diversion of water from the area might result in a lowered water table and fewer springs and this in turn would reduce stream discharge and warm surface water temperatures in the summer.

#### Waterfowl in Southeastern Michigan

Lake Erie, the Lower Detroit River, and Lake St. Clair are important waterfowl habitats. At least 25 species use the area with the largest concentrations occurring during the spring and fall migrations. There is an estimated 37.5 million days of bird use each year. The two most important areas are Anchor Bay in Lake St. Clair and the Lower Detroit River and adjacent waters of Lake Erie.

The marsh areas as well as the zone in the Lake less than 12 feet deep comprised most of the habitat that the birds depend on for food. Wildcelery and mollusks are the most important foods especially for black ducks, mallards, canvasback, redhead, and scaup. Food does seem to be the most important factor influencing



distribution in the area.

Mortality in the winter from oil pollution, starvation and disease have been well documented. Hunting is a major mortality factor in the fall. An average of 12,500 hunters harvest 75,300 ducks. This is about 11% of the ducks that use this area in the fall. The area serves 16.8% of the Michigan waterfowl hunters and 16.8% of the birds harvested in Michigan. Wastewater management plans should consider waterfowl habitats since this area does serve a significant number of birds which forms the base for a traditional fall outdoor activity.

#### Heavy Metals in Southeastern Michigan

Twenty-one heavy metals which appear to pose the greatest pollutional threat in the southeastern region of Michigan have been reviewed. Depending on their concentrations, all of these heavy metals can exhibit toxic effects to biota in an aquatic ecosystem. Eight of these elements; chromium, manganese, iron, cobalt, copper, zinc, molybdenum and selenium are required in trace amounts for the normal nutrition of either plants or animals. In addition, some data indicate that vanadium may also be an essential element. The remaining 12 of these elements are nickel, beryllium, antimony, tin, cadmium, lead, arsenic, mercury, titanium, aluminum, barium, and silver.

Heavy metals enter the aquatic environment of southeastern Michigan from the following sources:

1. Natural weathering of soluble heavy metal species from geological formations.
2. Point discharges of heavy metals from industrial operations.
3. Discharge of heavy metals from the sewage treatment plants.
4. Atmospheric fallout of heavy metals resulting from man's industrial, urban, and recreational activities.

Most of the industrial point discharges of heavy metals have been identified and tabulated in southeastern Michigan. Furthermore,

heavy metal contributions from sewage treatment plants and atmospheric fallout have been estimated. However, the tabulated data do not include heavy metal contributions from Ontario where the waters are common to both countries.

Heavy metals are normally removed from the water column and deposited in the sediments by adsorption and precipitation. How they are further partitioned in the sediments is not known. However, they appear to be most commonly associated with the organic fraction of the sediments. Some heavy metals (vis mercury and copper) have been reported to be almost quantitatively partitioned into the organic fraction of the sediments. Thus, the organic fractions of sediments appear to act as heavy metal reservoirs. As the physical, biological, and chemical conditions in the aquatic system change, these organically bound heavy metals can be released and recycled into the water column.

The amounts of heavy metals which are toxic to aquatic organisms cannot be accurately predicted at the present time. Therefore, toxicological data obtained from laboratory bioassay experiments must be cautiously applied to determine heavy metal concentrations that are "safe" for aquatic organisms in a natural water system. It has been suggested that the "safe" concentration for each heavy metal may be more realistically determined by first establishing the 96-hour  $LC_{50}$  for the most sensitive aquatic organism of that system and then applying a "safety factor" ranging from 0.1 to 0.01 depending on the persistence and cumulative effect of the toxicant on aquatic animals. But even this approach has serious drawbacks.

Heavy metals in natural waters also are a concern from a public health standpoint. The United States Public Health Service has enforced mandatory limits on the amount of heavy metals in public water supplies to protect the general public from this hazard. Therefore, except in the immediate vicinity of industrial or sewage treatment plant outfalls, the concentrations of heavy metals are within those limits in the surface waters of South-eastern Michigan.

### Heavy Metal Plant Interactions

Ca, Co, Cr, Cu, Fe, K, Mg, Mn, Mo, Se, Vn and Zn are among the elements that are essential for normal growth and reproduction of plants. They are associated primarily with enzymes that control the balance of vital processes and are required in very small amounts. Ag, Be, Hg, Ni, Pb and Sn are among the elements thought not to be essential for plant growth. They are highly toxic and have their effect by altering enzyme function.

Heavy metals must be available from the environment and get into the protoplasm of cells in order to effect plant communities. Plants can absorb levels of heavy metals that are not tolerated by mammals without apparent toxic effect on the plant. This has been assigned to storage in non-living portions of the plant cell such that the protoplast is shielded from their effect. Additions of heavy metals to the available environmental pool may promote growth in systems in which the pool is low, cause luxury consumption storage without effecting yield where the pool has intermediate concentrations, or be toxic and lethal in systems where the available pool is high.

Both increased production and selection for metal tolerant species upon addition of heavy metals to an ecosystem will favor a shift in speciation in resident plant communities and a lowering of diversity. Because levels of tolerance to heavy metals are ostensibly higher in plants than in animals, selective pressure on animal populations that use enriched plants will provide a feedback influencing the nature of the former.

Plants are able to concentrate heavy metals from their environment by a factor on the order of  $3 \times 10^3$ . In a broad range below toxicity, metals are absorbed in increasing quantity as their abundance is increased in the environment. The maximum difference in concentration across this range for submersed aquatic plants is one order of magnitude.

Planktonic algae and floating hydrophytes concentrate metals primarily from the water; benthic algae and submersed vascular hydrophytes concentrate metals from the sediments and ambient

water, the latter having a disproportionately greater impact on tissue levels in hypereutrophic environments; emergent aquatic plants concentrate metals primarily from the hydrosol; terrestrial vegetation that is not spray irrigated concentrates elements from an aerated soil.

Environmental factors influencing the concentration of heavy metals by plant tissues are (a) form of the metal, (b) ion interactions, (c) amount of freely diffusible metal in the vicinity of uptake structures, (d) oxygen concentration, and (e) the rate of flow of metals past absorbing tissues. Plant mechanisms influencing the concentration of heavy metals by plant tissues are (a) a relative selectivity for different metals that are in the environment, (b) the rate of metal loss by excretion to the environment, and (c) proportion of uptake by roots vs. foliage in submersed aquatic plants.

Temperature limits growth and heavy metal uptake in plant populations of Lake Clair to the April 1 - November 1 period. If heavy metals (or other materials) occur in the Lake St. Clair basin at toxic concentrations, conditions of oxygen and light that are intolerable or marginal for plant growth will likely exist there as well. Metal cycles in these regions will not be strongly influenced by plant populations.

Areas of highest available heavy metal concentrations are likely to exist near the outer boundary of stream flow and more lentic water of the lake. Production of phytoplankton should be promoted in this region, rather than macrophytes, because of turbidity and siltation considerations. Metals absorbed here by the phytoplankters will be transported and deposited in quiet portions of the basin as the algae sink and decompose. The most pronounced effect of plant populations on the trace metal dynamics of the system is likely to be an autumnal pulse of metals caused by the dispersion of metal-holding fragments of plant decomposition. Although data on the heavy metal content of the macrophytes in the Lake St. Clair - Lake Erie system was not available, estimates have been provided from the information on Chowla and Chau (1969)



and Bowen (1966) and Chapman, et al. (1968).

#### Pesticides and Synthetic Organics in Southeastern Michigan

The southeastern region of Michigan is subject to chemical contamination from agricultural, industrial and domestic wastes. Monitoring efforts to the present time have been concerned primarily with organochlorine compounds in aquatic systems. DDT and dieldrin in relatively low concentrations are ubiquitous in water and fish from this region. Most data indicates a general distribution of DDT with more localized sources of dieldrin contamination. The concentration of these pesticides are below levels known to be acutely toxic in biological systems, but they do present possible hazards to animals in higher trophic levels (e.g. birds). However, to the present time there is no evidence that DDT or dieldrin have caused significant damage to either terrestrial or aquatic communities in southeastern Michigan. A decline in DDT and dieldrin residues is expected to occur as the use of these compounds declines in favor of other pesticides.

Less effort has been given to detection and quantification of other pesticides in the region. Pesticides such as chlordane that have residual activity in soils and water may present hazards similar to DDT and dieldrin if concentrations accumulate in soils or in the aquatic biota. Some less persistent and less toxic pesticides that are repeatedly used over extensive areas of agricultural land (e.g. atrazine) should be evaluated for their potential contamination and hazard to both terrestrial and aquatic systems. Very little is known of interactions of chemicals in the environment. Biological communities are insulted with many different organic chemicals that have variable toxicity and persistence. Although few examples of synergism or potentiation are known to occur in the environment, this possibility does exist. Several studies have shown that residue accumulation of one compound may be either increased or reduced by exposure to a second compound.



Industrial and domestic wastes may pose a greater hazard in southeastern Michigan than contamination from pesticides. Present data shows polychlorinated biphenyls are widely distributed in the aquatic environment. Phthalic acid esters are also known to occur in wastewater effluents. Much more information is needed to determine if these compounds are causing damage to biological communities.

The few organic compounds that have been monitored in the southeastern Michigan region represent a small fraction of those that have been found in other regions of the country. A listing of organic chemical pollutants found in various aquatic systems is compiled in the Environmental Protection Agency's Water Quality Criteria Data Book, Volume 1 (1970). This reference should be consulted to determine the relative acute and chronic toxicity of various organic chemicals that occur in wastewater and to set priorities for environmental monitoring. Industrial wastes, final wastewater effluents and sewage sludge should be monitored for highly toxic or persistent compounds.

The problems presented by toxic organic chemicals must be considered in any wastewater treatment plan. Industrial discharges that contain toxic chemicals should be treated before entering municipal wastewater treatment plants. To prevent general environmental contamination treatment should occur at the point of greatest concentration (i.e. component wastes within the industry or the total industrial effluent).

Dilution will not solve waste problems where highly persistent compounds are concerned. The great potential for residue accumulation in higher trophic levels prevents effective waste dilution. Residues in wastewater that is distributed over land areas (e.g. spray irrigation) may accumulate to hazardous levels in the soil and eventually reenter surface waters. Crops and terrestrial wildlife may be damaged by the long term accumulation in the soil. Sewage sludge presents similar problems in land disposal.

Ideally, toxic compounds which are persistent should be removed from industrial wastes before entering municipal wastewater

treatment plants. The isolation and destruction of these compounds will prevent damage to broad areas of the environment.

## SECTION VI - ECOLOGICAL IMPACTS

### General

The impact zone for waste management practices in southeastern Michigan as stated earlier in this report encompasses not only the southeastern Michigan area and its adjacent waters but a much larger area extending throughout the entire Lake Erie basin. It was also indicated earlier that the contribution of pollutants from southeastern Michigan area is very large, and in fact, constitutes the major source to the basin. Considering the above it was initially thought that implementation of a major wastewater treatment system for southeastern Michigan would, in itself, create significant improvements in water quality for Lake Erie, even if all other inputs continued at their present rates. On the basis of our baseline studies it is now fairly safe to conclude that even a 100% elimination of municipal and industrial wastes from southeast Michigan coupled with clean up of Michigan's tributary streams would not be adequate in significantly improving conditions in Lake Erie. In terms of phosphorus inputs to the western basin, the tributary streams from southwestern Ontario and the Canadian municipal and industrial inputs plus the loading from the Maumee River (Table 43) total about 10 million pounds of phosphorus per year. This loading coupled with over 6 million pounds of phosphorus per year introduced from Lake Huron is more than enough to sustain the high level of eutrophy in the western basin. Thus, for any major improvements of water quality to be realized in Lake Erie it is essential to reduce inputs from all watersheds bordering the Lake; not just Michigan's. The following discussion elaborates what we believe to be one of the most critical factors governing the trophic status of Lake Erie; phosphorus. This discussion provides the rationale for many of our conclusions on the environmental impact of wastewater management strategies.

Table 43. Synthesized phosphorus loadings to western Lake Erie.

Source	Phosphorus Loading - lbs/yr
Lake Huron	Total 6,189,000
Detroit River (excluding Lake Huron)	
U.S. Municipal and Industrial	18,929,000
U.S. Runoff	1,066,000
Canadian Municipal and Industrial	2,946,000
Canadian Runoff	1,369,000
	Total 24,310,000
Maumee River	
Municipal and Industrial	3,448,000
Runoff	1,777,000
	Total 5,225,000
Huron River	
Municipal and Industrial	257,000
Runoff	165,000
	Total 422,000
Raisin River	
Municipal and Industrial	187,000
Runoff	210,000
	Total 397,000
Portage River	
Municipal and Industrial	154,000
Runoff	174,000
	Total 328,000
-----	
GRAND TOTAL	36,871,000

### Recovery of Lake Erie Through Phosphorus Control

Phosphorus removal from wastes has been suggested as the single most outstanding treatment need for Lake Erie after direct disease threats are eradicated. To our knowledge, no concise and specific prediction of the response to phosphorus removal within the Lake Erie watershed has been published. The reason is simple; the recovery process is very complicated and highly hypothetical and we can only consider the likelihood of alternative possibilities in the light of the available data (which are frequently questionable). Total response within the lake not only depends on the phosphorus removal but the specificity of the process used to remove the phosphorus. Some tertiary treatment processes, land disposal for example, are likely to remove much more than phosphorus alone, and conceivably, a favorable response in receiving waters may be due to more than phosphorus removal alone.

It is doubtful that any single waste treatment process will achieve 100% phosphorus removal, but for the sake of illustration consider the impact of 100% removal of phosphorus within a hypothetical treatment scheme for southeastern Michigan. Further, assume that the removal is confined to municipal, industrial and storm discharges. After the phosphorus removal from these sources, this element can enter via Lake Huron, tributary streams outside the boundaries of southeastern Michigan, small atmospheric inputs and from existing sediments within the lake proper. In total these inputs to the western basin approximate 16 million pounds of phosphorus per year. If we can optimistically assume that the concentration of phosphorus in the western basin is a linear function of the annual phosphorus loading, then 16 million pounds of phosphorus per year would yield a concentration of approximately 45  $\mu\text{gP/liter}$  or approximately twice the concentration that would limit algal abundance.

Once in awhile, phosphorus may become limiting in the western basin when other limiting agents are mitigated, and an algal bloom follows. Occasionally something happens in the western basin that is associated with warm calm (but not necessarily stratified) periods and



allows nearly all phosphorus and nitrogen in the water to be incorporated by phytoplankton with an associated order of magnitude increase in planktonic carbon. Otherwise only one third of the phosphorus present is required to maintain the average concentration of algae in the western basin. Whatever regulates the occurrence of blooms is unimportant. Blooms would not occur if phosphorus were unavailable.

Within the western basin (again assuming a linear relationship between phosphorus loading and concentration) a 75 to 80% reduction in total phosphorus inputs would be necessary to reduce the western basin concentration to 20-25  $\mu\text{g/l}$ . Existing western basin concentrations are approximately 100  $\mu\text{g/l}$  (See Table 44). To achieve this would require at least 90% phosphorus removal from all municipal, industrial and storm discharges to the western basin plus reduction of phosphorus from streams, such that every stream entering the western basin carried a base flow concentration of 30  $\mu\text{g/l}$  (a ten-fold reduction over existing levels). With sound land management practices and existing waste treatment technology this reduction is probably within reach.

Considering the total basin, complete removal from wastes from southeast Michigan will account for about 50% of all phosphorus entering Lake Erie (IJC, 1969). Even if all phosphorus were removed from all municipal and industrial wastes that enter the lake, about 25% would not be included because of remaining watershed sources. Total nitrogen would be even less limited because a much larger proportion of that element is thought to come from non-point sources in the watershed. If sediments contributed little, complete treatment for southeast Michigan alone would probably influence the frequency and intensity of algal blooms but average algal concentrations and, less certainly, the usual species composition, would not be as likely to change. If 100% removal of all industrial and municipal input were instituted and sediments contributed little, the mean phytoplankton concentration would probably decrease slightly in the western basin and blooms would be much less likely to occur.

Table 44. Nutrient concentrations in sampling regions 3, 5, 7 for period 1969-1971.

Sampling				Parameter <sup>2</sup>							
Station	Year	Month <sup>1</sup>	No. of Samples	1	2	3	4	5	6	7	8
SOUTHERN LAKE HURON:											
7E1	1969	7	1	0.13			0.2		0.14	0.003	0.003
	1970	7	1				0.1			0.002	0.003
7E2	1969	7	1	0.08			0.2		0.15	0.003	0.003
	1970	7	1				0.1			0.001	0.004
7E3	1969	7	1	0.07			0.2		0.12	0.003	0.003
	1970	7	1				0.2			0.002	0.003
7E4	1969	7	1	0.10			0.3		0.16	0.003	0.004
	1970	7	1				0.1			0.001	0.003
7E5	1969	7	1	0.07			0.3		0.16	0.003	0.003
	1970	7	1				0.1			0.001	0.003
7E6	1969	7	1	0.08			0.3		0.17	0.001	0.004
	1970	7	1				0.1			0.001	0.004
7E7	1969	7	1	0.12			0.2		0.38	0.020	0.002
	1970	7	1				0.1			0.001	0.004
7E8	1969	7	1	0.11			0.2		0.84	0.003	0.004
	1970	7	1				0.2			0.001	0.003
7E9	1969	7	1	0.18			0.3		0.19	0.003	0.003
	1970	7	1				0.2			0.001	0.003
7E10	1969	7	1	0.08			0.5		0.49	0.003	0.003
	1970	7	1				0.1			0.001	0.002
7E11	1969	7	1	0.12			0.3		0.50	0.003	0.004
	1970	7	1				0.2			0.003	0.010

Table 44. (con't.)

LAKE ST. CLAIR:									
5E1	1969	7, 10	2	0.09	0.15	0.25	0.007	0.024	
	1971	8	1	0.12	4.4	0.34	0.22	0.010	
5E2	1969	7, 10	2	0.08	0.20	0.20	0.017	0.030	
5E3	1969	7, 10	2	0.07	0.20	0.23	0.003	0.015	
	1971	8	1	0.23	4.6	0.43	0.20	0.010	
5E4	1969	7, 10	2	0.10	0.10	0.31	0.120	0.145	
	1971	8	1	0.16	5.1	0.67	0.51	0.200	
5E5	1969	7, 10	2	0.12	0.15	0.42	0.070	0.105	
	1971	8	1	0.26	5.2	0.64	0.38	0.090	
5E6	1969	7, 10	2	0.07	0.15	0.16	0.027	0.035	
	1971	8	1	0.29	4.4	0.38	0.09	0.010	
5E7	1969	7, 10	2	0.09	0.20	0.16	0.003	0.009	
	1971	8	1	0.27	4.4	0.45	0.18	0.010	
5E8	1969	7, 10	2	0.14	0.20	0.45	0.120	0.175	
	1971	8	1	0.48	5.8	1.10	0.58	0.300	
5E9	1969	7, 10	2	0.23	0.20	0.30	0.070	0.105	
	1971	8	1	0.23	4.7	0.38	0.15	0.020	
5E10	1969	7, 10	2	0.10	0.15	0.22	0.005	0.019	
5E12	1969	8	1	0.10	0.15	0.19	0.032	0.044	
5E13	1969	7, 10	2	0.08	0.15	0.26	0.023	0.045	
	1971	8	1	0.17	4.5	0.51	0.34	0.020	
5E14	1969	7, 10	2	0.09	0.15	0.23	0.013	0.025	
	1971	8	1	0.15	5.0	0.39	0.24	0.020	
WESTERN LAKE ERIE:									
3E7	1969	7, 9, 10	4	9.6	105	0.35	0.060	0.107	
			3	0.28	0.7		0.040	0.100	
3E8	1970	8, 9	2	8.1	96		0.045	0.105	
	1970	8, 9	2	7.7	90				

Table 44.(con't.)

3E9	1970	8,9	2		7.1	83	0.1		0.040	0.095
3E10	1969	7,9,10	4		9.9	107				
			3	0.29			0.2	0.32	0.073	0.113
3E12	1970	8,9	2		7.3	85	0.2		0.070	0.210
	1969	7,9,10	4		8.1	86				
			3	0.24			0.2	0.33	0.060	0.090
3E13	1970	8,9	2		7.1	82	0.2		0.055	0.130
3E17	1970	8,9	2		6.7	76	0.3		0.090	0.180
3E32	1969	7,9,10	4		6.3	71	0.3		0.085	0.170
			3	0.26	9.3	101				
3E33	1969	7,9,10	4				0.7	0.35	0.057	0.093
			3		9.7	106				
3M14	1969	5,6,8,9	12	0.30			0.7	0.39	0.050	0.100
	1970	6,7,10	9	0.47	6.5		0.14			0.37
3M15	1969	5,6,8,9	8	0.30	6.7		0.31			0.24
	1970	7,9,10	8	0.45	6.6		0.15			0.32
3M16	1969	5,6,8,9	16	0.36	6.7		0.32			0.24
	1970	7,9,10	16	0.33	6.9		0.10			0.19
3M18	1969	5,6,8,9	12	0.34	7.0		0.24			0.21
	1970	7,9,10	12	0.14	7.6		0.10			0.13
			12	0.11	7.3		0.20			0.14

<sup>1</sup>1 (January), . . . , 12 (December)

<sup>2</sup>Parameters:

1. Ammonia (NH<sub>3</sub>), N, mg/l
2. DO, mg/l
3. DO, saturation, %
4. Nitrate (NO<sub>3</sub>), N, mg/l
5. Nitrogen - total Kjeldahl, N, mg/l
6. Organic Nitrogen, N, mg/l
7. Phosphorus - dissolved, wet method, P, mg/l
8. Phosphorus - total, wet method, P, mg/l

### *Role of Sediments:*

Conservative evaluation of sediments as a potential source of phosphorus could underestimate sustained levels of phosphorus following treatment, especially in the central basin where nearly anaerobic bottom waters are found each summer. Within the upper 10 cm of sediment, phosphorus is frequently resuspended, at least in the western basin. At least a part of the resuspended phosphorus may find its way back to algae and eventually much of it could end up in the central basin where, under anaerobic conditions, soluble phosphorus is likely to be returned to surface waters. Even if hypolimnetic anoxia remained the same, phosphorus concentrations from sediments would eventually dissipate. The rate of dissipation will depend primarily on the rate of oxygen demanding organic fallout into the hypolimnion (which depends on primary productivity) and the amount of phosphorus present in the upper sediments. If organic fallout were cut drastically oxygen depletion rates in the hypolimnion of the central and eastern basin might be forestalled. However, there is some meager evidence that oxygen depletion to nearly an anaerobic state has existed in the west-central basin since 1930 at least (Wright, *et al.*, 1955). Standing crops of phytoplankton (Davis, 1966) and probably primary productivity, have increased by three times in the central basin during that period. We might reasonably expect to return to those levels found in 1930 in the eastern part of the central basin if phosphorus levels are cut back to at least 1/3 of present mean concentrations in overlying water (10 - 15  $\mu\text{g/liter}$ ). In the eastern basin this would entail a cut back by about half of present concentrations.

### *Basin Interactions:*

There is presently a progressive mean decrease in total phosphorus concentration of about 60-70% as water passes from the western to the eastern basin. The difference ends up in the sediments. If sediments could be ignored, a 50% reduction of phosphorus input to the western basin could result in up to a 66%



removal of phosphorus from water in the eastern half of the lake. However, as aquatic concentrations decrease, the net flux from the sediments may severely dampen the effects of removal.

An empirical model presented by Thon (1969) suggest that the eastern basin outlet concentration will be halved by decreasing the total phosphorus input by about 85% rather than by the 50% reduction inferred without reference to the effect of sediments. Thus, during early recovery, the sediment source of phosphorus depresses the response of aquatic phosphorus to input control by about 40% (for every 5 pounds removed at the source 2 pounds will be returned to the water from the sediments). But because phosphorus seems to be presently limiting in the eastern basin any decrease in phosphorus inputs to the lake should have an impact on organic synthesis and hypolimnetic oxygen demand in the eastern part of the lake. Thus, removal of all municipal and industrial phosphorus entering the lake has that much greater chance of decreasing plankton biomass and oxygen depletion in the eastern half of the lake. It appears, however, that this phosphorus reduction may not necessarily restrict productivity and potential  $O_2$  depletion in the western half of the lake. Land runoff and perhaps, atmospheric inputs, would have to be controlled as well.

Making predictions on the clearance rate of phosphorus from the waters of Lake Erie depends on the continuity of oxygen depletion. In the western basin oxygen depletion is in total probably of minor importance in the recycling of sedimentary phosphorus to overlying waters. However, resuspension of sediments is most important in the western basin and even though this occurs in a well oxygenated environment with high alkalinity and iron concentrations, some phosphorus from resuspended anaerobic sediments may become available for photosynthesis. Just how much is unknown. If no great change in organic synthesis occurs as a result of control of phosphorus input from point sources in the western basin, organic export to the central basin will continue. That export would contribute to a continuous  $O_2$  depletion just east of the islands whenever a thermocline formed. Incomplete control of

phosphorous from the Cleveland area would serve to aggravate this problem. There is some indication from early reports of Whitefish movements (Regier, *et al.*, 1970) and oxygen concentrations in 1930 (Wright, *et al.*, 1955) that the west-central region underwent some  $O_2$  depletion before cultural enrichment became influential. Yet algal biomass has obviously increased since 1930 primarily because of the extension of spring and fall blooms (Davis, 1966). That increase, by way of extenuation of biomass through the summer months, implies a growing continual source of limiting nutrient in the epilimnion; probably phosphorus. There is no direct evidence that the phosphorus input rate via the resuspension of western basin sediments has increased, but it probably has in proportion to inputs.

Undoubtedly, inputs from the Cleveland area have increased since 1930 and this may be the major source of soluble phosphorus for the west-central basin. Absolute limitation of phosphorus input from the Cleveland area as well as from sources around the western basin would probably depress mid-summer algal populations but have little influence on spring and fall blooms that probably occur following mixture of enriched hypolimnetic waters with nutrient depleted epilimnetic waters. Data has been gathered by FWPCA (1968) that demonstrates the nutrient enrichment of the central basin hypolimnion. This hypolimnetic enrichment will likely continue indefinitely accompanied by algal blooms, that are associated with destratification. The intensity of blooms may be depressed in the future by tight regulation of municipal and industrial wastes but nutrient budgets will probably equilibrate to a point similar to those reported by Davis in the 1920's when seasonal blooms were very noticeable but temporally restricted to a few weeks. It is probable that recovery to conditions that are similar to those in the 1920's could come about within a very few years.

### *Biological Effects:*

Effective control of phosphorus through any technique will also remove organic wastes and bacterial contamination although viral contaminants may be more resistant. Therefore, those shoreline areas presently contaminated by municipal and some industrial organic waste (slaughter houses for example) will recover from present oxygen demanding conditions very rapidly. However, these effects alone may bring about only limited changes in lake biology because the impact of those wastes upon the lake appears to be minor compared to the organics generated by photosynthesis. In recent years at least (and probably earlier too) oxygen depletion in the western basin during temporary stratification has occurred generally throughout the basin and far from the direct influence of the Detroit (usually fairly well oxygenated) and Maumee Rivers. Yet benthic distributions indicate that changes in the rivers have apparently had the greatest impact on the benthos, changes that apparently have extended effects into the lake about 10 times greater than can be explained by increases in simple biological oxygen demand. If zoobenthic changes are associated with environmental toxins of various kinds, any recovery of zoobenthos is questionable without removal of toxic inputs.

Although the marsh lands bordering Lake Erie, Lower Detroit River, and Lake St. Clair have been reduced to a fraction of the former and habitat deterioration from various forms of pollution has occurred, significant numbers of waterfowl use the area. Improved wastewater management programs will be beneficial to waterfowl by controlling toxic substances such as oils, heavy metals, and biocides that may cause direct mortality and by maintaining the plant and animal communities that serve as food.

Changes in the fish population are probably much more dependent now on the management of fisheries in Lake Erie than on some reversal of eutrophication. The extinction of the bluepike and introduction of successful exotic species make it unlikely that the fishery will ever be restructured to what it was fifty years

ago, although whitefish and walleye may again become relatively important species if managed appropriately. Recovery to conditions that were described for the 1800's are impractical if not impossible to attain. The existence of those large standing crops of fish often comprised of slowly maturing fish was apparently dependent on extensive beds of aquatic macrophytes, low turbidities, coarser sediments, and stable spawning tributaries free of suffocating and blinding sediments, dams and other structural and polluttional impasses. Only very extensive modifications of land use throughout the watershed will allow any semblence of complete recovery.

#### Recovery of Southeastern Michigan Tributary Streams

In all cases studied, southeastern Michigan tributaries had the worst ecological conditions below municipal or industrial outfalls, but reasonable biological populations existed in other parts of the stream. The biota in sections downstream from the recovery areas of point sources must still contend with the non-point input of agriculture and runoff from natural areas. With the exception of sediment input and some possible warming from surface runoff, the nutrients added from non-point sources seems relatively insignificant. Thus with an overall upgrading of water quality which assumes nearly complete removal of BOD and efficient (95%) phosphate control the following changes should occur in the streams: The Black, Pine and Belle Rivers would show some improvement largely in the lower reaches. The upper half or third of these streams would be trout water having many intolerant invertebrate forms. The lower parts would be smallmouth bass and northern pike water.

The headwaters of the Clinton, the Huron and the Rouge Rivers already contain trout. This trout fishery is maintained by stocking and it is likely that it would have to continue to be stocked; however, the water suitable for fingerling and adult trout would expand considerably. Likewise, downstream populations of intolerant invertebrates would develop rapidly, being colonized

by drifting organisms from upstream. It is likely that the lower reaches of these streams could be stocked with walleye fingerlings and once again have this high quality food and game fish. Northern pike and smallmouth bass would reproduce naturally. The lower part of the Detroit River would develop fish populations much like the upper Detroit River.

Perhaps the greatest improvements would be possible and would occur in the streams in the worst ecological condition at present. Thus the Raisin and Maumee Rivers have the greatest potential for recovery. While the headwaters of these streams may never support trout populations they would be excellent smallmouth bass streams. The lower Maumee already contains walleye and it is probable that both streams could have walleye fisheries, but these may have to be maintained by the stocking of fingerlings.

Low oxygen concentrations in the streams would be less likely to occur. Some diurnal sags would probably still be present during summer periods, but would not be serious. Primary production would be reduced greatly in all streams but the St. Clair, Detroit, and Maumee Rivers. This would occur with effective phosphate removal. While nitrates are probably limiting in the other streams presently it is only for the diatoms and green algae; the blue-greens being able to fix their own nitrogen in many cases.

Since the St. Clair and Detroit Rivers receive phytoplankton produced from the lakes, primary production may change little in those streams. Exceptional phosphate removal will be necessary to reduce the concentrations of the Maumee to limiting levels. This again may be true only for the situation at the stream mouth below Toledo.

The concentration of nitrates in the Raisin and Maumee are very high and may receive considerable input from agriculture since both streams drain extensive farm areas. The high levels of nitrates may continue, but would be ineffective in producing large algal crops if the phosphate can be controlled.

While there would be great improvement in the overall species composition of the streams, there would probably be a decline in



the total pounds of fish produced in the streams. Removal of nutrients would cause this decline in production. However, the decline would be in rough fish production while production of game fish would be increased over present crops.

Streams in agricultural or urbanized areas are unstable in discharge when compared to streams with natural watersheds. In other words, the increase in surface or storm runoff facilitated by plowed or paved surfaces causes floods during precipitation and low flows at other times since the water does not move through the ground and out as springs. The loss of springs also causes a general expansion of water temperatures, warmer in the summer and colder in the winter. Floods and plowed fields also contribute greatly to the sediment problem.

It is likely that the inland streams (not the St. Clair or Detroit) will continue to be unstable. Agriculture and floods will continue to contribute to the sediment problem. Sedimentation is often the main reason for aborted spawning attempts by smallmouth bass and stream dwelling walleyes. Thus, while the streams may support adults, natural reproduction may have to be supplemented by stocking programs. Even more critical, invertebrate populations can be smothered by heavy silt loads. However, present invertebrate populations seem adequate as a food base.

Treatment methods that would centralize wastes and discharge directly into the Great Lakes would be most effective in removal of nutrients and BOD from the streams. However, this process would also remove much of the "base" flow of the streams. If the water supply is from ground water sources then this flow probably occurred before development as springs from a higher water table. Removal would deprive the channel of water normally required to fill pools and maintain riffles. While reduced flow may not harm fish species accustomed to small streams, it may reduce the probability of establishing larger stream forms such as walleyes and channel catfish.

Land disposal methods, on the other hand, may effectively serve to raise the water table and cause an increase flow of springs

even though the land is tiled. Tiled drainage and springs would both result in maintaining the original base flow of the streams.

#### Effects of Proposed Wastewater Management Alternatives on Lake Erie and Southeastern Michigan Tributary Streams

The following alternatives were examined: (1) Existing State of Michigan Plan; (2) Advanced Biological Wastewater Treatment (AWT), Total; (3) Physical Chemical (P-C) Treatment, Total; (4) Land Disposal, Total; (5) AWT-PC Combined; (6) Land for Municipal and Industrial, PC for Storm; (7) Extended AWT; and (8) Extended Land.

#### *Stream Effects:*

The impact of all of the above alternatives on southeastern Michigan tributary streams is expected to be large and in a favorable direction. While some of the northernmost streams are in fair condition at the present time most of the others have suffered severe degradation. As was discussed earlier, the degradation was due, in large part, to discharge of municipal and industrial waste. With the overall increase in water quality that can be expected in all eight alternatives it is very likely that the intolerant benthic fauna and game fish would expand throughout the length of all southeastern Michigan streams. The upper third of the northern group of streams should be of such quality as to support populations of trout, although they may have to be maintained by periodic stocking. The lower reaches of the Raisin, Huron, Rouge and Clinton should demonstrate very significant improvements in water quality and will probably support good populations of smallmouth bass. Walleye may be maintained in the Raisin River by a stocking program.

Because of cultural modifications of the watersheds due to urbanization and agriculture the streams cannot realistically be expected to return to their original condition. Unintercepted runoff from agricultural land will continue to introduce silt, nutrients and toxic materials such as pesticides. This impact

is very difficult to ascertain with the data at hand since the agricultural effects are probably masked by effects of municipal and industrial wastes.

To rank each of the eight alternatives in terms of their effectiveness in improving stream conditions would be very difficult for the degree of resolution between each is not great. We can, however, identify groups of alternatives that appear better in overall effectiveness. These are shown below:

More Effective



Less Effective

Land Disposal - Total  
Land - Municipal and Industrial,  
P-C for Storm

Extended Land  
Extended AWT

AWT Total  
P-C Total  
P-C - AWT Combined

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The above ranking is based on treatment efficiencies for nutrient removal and efficiencies for removal of toxic materials. Land disposal is expected to perform best according to this criteria and as such ranks the highest. Total treatment with AWT or P-C will also be very effective in nutrient removal but perform somewhat lower in terms of removing pesticides, synthetic organics and heavy metals. The proposed State of Michigan plan while a great improvement over existing processes is somewhat less effective in nutrient removal and is thus ranked as the least effective of the group. Again, the differences in effectiveness between all of the alternatives is not great and major stream improvements can be expected under any of the alternatives.

### Lake Effects:

The values in Table 43 (p. 225) serve as a focal point for our lake assessments. These values represent a synthesis of our computed loadings and those published by the IJC (1969). Of particular importance in this table is the municipal and industrial contribution of phosphorus to the Detroit River. This value can either be calculated directly by summing all point source inputs or arrived at indirectly. The precision of our calculated loadings were such that we placed the most confidence in the indirect approach. The loading at the mouth of the Detroit River appears to be a very accurate estimate based on findings of IJC (1969) and Harlow (1966). Their estimates were 30,720,000 and 30,656,000 lb P/yr respectively. Our independent estimate was 30,499,000 lb P/yr which for all practical purposes is identical. Our estimates for tributary stream loadings into the Detroit River, however is significantly lower than those calculated by IJC (1969). We believe our estimates to be more accurate since they encompass not only a three year period but also consider a larger cross section of sampling agencies and number of samples. Our calculated phosphorus loading from Lake Huron is larger than the IJC (1969) estimate, but again we believe our estimate to be a more accurate representation for the same reasons. Thus, we arrived at the municipal and industrial loading by subtracting the Lake Huron and tributary loading from the lower Detroit River loading. This subtraction results in a value of 21,875,000 lb P/yr from municipal and industrial sources. This value, then, represents the controllable phosphorus input from the Detroit River.

We may achieve further control by reducing phosphorus inputs upstream from the tributary stream mouths but this figure is difficult to calculate. We can, however, make a reasonable assumption that if sound land management and state-of-the-art waste treatment is practiced on the miscellaneous upstream phosphorus sources the final phosphorus concentration at the stream mouths would be approximately 30  $\mu\text{g/l}$  or better. Using this concentration we can recalculate a "base flow" loading for all the

tributary streams. This value is approximately 600,000 lb P/yr for all tributary streams into the western basin.

To achieve phosphorus limitation for the western basin (20-25  $\mu\text{g P/l}$ ) means that a 75-80% reduction in all phosphorus inputs is necessary. The loading to achieve this condition ranges from 7,300,000 to 9,200,000 lb P/yr (See earlier discussions on limiting nutrients p. 226-231). Subtracting the Lake Huron and tributary stream base loadings gives an allowable input of 500,000 to 2,400,000 lb P/yr to stay within the limits of the total loading necessary to limit phosphorus.

Each of the eight proposed alternatives was assessed in terms of its effect on phosphorus loading to see if the maximum allowable loading of approximately 2,400,000 lb P/yr would be exceeded. This assessment is summarized in Table 45. As can be seen the phosphorus loading resulting from the Michigan plan will significantly exceed this limit. The total AWT and P-C plan proposed by the Corps of Engineers will result in a loading of 1,500,000 lb P/yr. The resulting phosphorus concentration in the western basin under these alternatives could conceivably approach 25  $\mu\text{g/l}$ . The total land disposal system (as well as land for municipal and industrial and P-C for storm) has a significantly lower loading and could well result in phosphorus limitations to the western basin. The above loading calculations considered both municipal and industrial flows and storm flows.

In summary, the most favorable impacts to the western basin will be realized under the land disposal alternative. In addition to the high effectiveness of nutrient removal under this alternative, heavy metals, pesticides and synthetic organics will also be more effectively removed. A quantitative assessment of removing these hazardous materials is not possible, but it is safe to conclude that the impact would be in a favorable direction.

The conclusions expressed above are highly dependent on the assumption that resultant phosphorus concentrations in the western basin are a linear function of the inputs. Since this is not entirely true as stated earlier, an 80% reduction in phosphorus input will create less than an 80% reduction in the western basin



Table 45. Summary of phosphorus loadings to the Western Basin of Lake Erie under proposed wastewater management alternatives.

Source	lb. Phosphorus per year
Lake Huron Input	6,200,000
<sup>1</sup> Stream Inputs	600,000
<sup>2</sup> Non-Michigan Municipal and Industrial Inputs	650,000
NON CONTROLLABLE LOADING	7,450,000
LOADING TO ACHIEVE A P CONCENTRATION OF 25 µg/l IN WESTERN BASIN	9,200,000
ALLOWABLE MUNICIPAL AND INDUSTRIAL LOADING FROM MICHIGAN	1,750,000
Loading with Michigan Plan	5,430,000
Loading with Corps AWT and PC-Total Plan	1,500,000
Loading with Corps Land Disposal Total Plan	500,000
<sup>1</sup> Assuming that the P concentration in all western basin tributaries is reduced to 30 µg/l.	
<sup>2</sup> Assuming a 90% phosphorus reduction in all non-Michigan municipal and industrial inputs to the western basin.	

concentration. Thus, even under the proposed land disposal alternatives the improvements to Lake Erie may not be as great as one might expect (at least for the first few years). It is likely that Lake Erie will not look any less "dirty looking" but swimming and other water contact sports would be safe throughout the lake. It is somewhat difficult to predict the effect on water treatment in the western basin because suspended solids are comprised most often of inorganic debris imported from disrupted watersheds and not particularly algae. The dominant genera of algae may change but all genera involved have been implicated with water taste problems. In the central and eastern basins, however, the abundances of undesirable algae could be reduced. Also in those basins inorganic debris is less important so decreases in algal abundance could reduce drinking water filtration costs more substantially. The production of cladophora, the major nuisance algae washed on beaches and attached to objects in water, may be reduced in the eastern basin but probably not eradicated. The expected change will be less for the central basin and even more so for the western basin.

Benthic recovery in the western basin may occur if unidentified toxins, phosphorus and intense BOD are removed from municipal and industrial effluents. If this is done, recovery of the burrowing mayfly to nuisance abundances is possible even though oxygen demand under stratified waters probably will remain high because of constant resuspension of sediment and continued high productivity. The response of fish species composition is difficult to predict. They appear to be more dependent on fishery management than on environmental manipulation at this level.

From the casual observers position, phosphorus control may do little for aesthetics because the shoreline in western Lake Erie will remain crowded with industry and other human activity and the water will remain extremely turbid. Dead fish will continue to show up on beaches as they do wherever they are abundant. In the central and eastern basins the water may become a little clearer and cladophora a little less abundant.

Ranking each of the eight alternatives in terms of their effectiveness in improving conditions in Lake Erie can be done much the same as was done for displaying the impacts on tributary streams. Unlike the streams, however, the lake system is not expected to demonstrate such large improvements. We believe that there exists a threshold concentration for phosphorus in the basin below which biologically significant changes would be expected to occur. Our best estimates of this concentration, based on published literature for other lakes of the world and on our own data, appears to be in the neighborhood of 20-25  $\mu\text{g P/l}$ . Only one of the proposed alternatives, the State of Michigan Plan, appears to result in a phosphorus loading that would definitely exceed the limits shown in Table 45. This plan would add 5,430,000 lbs. phosphorus per year, 3,680,000 lbs. phosphorus per year more than the upper limit of our calculated allowable limit of 1,750,000 lbs. phosphorus per year.

The other seven alternatives are much more difficult to assess. Even though Land Disposal appears to be well within the allowable limit in terms of phosphorus loading, sediment interactions discussed earlier could reduce its effectiveness. Advanced Waste Treatment and Physical Chemical Treatment also appear to be effective but can be considered borderline. Below is our ranking of the eight alternatives in terms of their probability of success in "cleaning up" Lake Erie.

Highest Probability  
of Improving Lake  
Erie



Lowest Probability  
of Improving Lake  
Erie

Land Disposal - Total  
Land - Municipal and Industrial  
P-C for Storm

Extended Land  
Extended AWT

AWT Total  
P-C Total  
P-C, AWT Combined

Michigan Plan

## Other Impacts of Proposed Wastewater Management Alternatives

### *Sludge:*

From an ecological standpoint there are two basic sludge handling practices proposed for the southeastern Michigan study. One is incineration and the other is land disposal. If sludge were simply composed of organic matter originally derived from photosynthesis and if oxidation were complete during incineration the by-products would be primarily harmless. Large amounts of  $\text{CO}_2$  and  $\text{H}_2\text{O}$  would be released to the atmosphere but these additions simply complete natural biogeochemical cycles. Proper engineering could insure small releases of particulate emissions and atmospheric contamination would be minimized. Releases of nitrogen oxides, sulfur oxides and small amounts of phosphorus will occur but their quantity and impacts have not been adequately evaluated.

Perhaps more important is the introduction of heavy metals to the atmosphere. As was indicated in the *Feasibility Phase* for the southeastern Michigan Study these materials are also included in sanitary sludges and many are in fact released to the atmosphere. The magnitude of these releases is almost completely unknown and their secondary impacts even more so. This remains as an important area for future research.

Land disposal of sludges is an alternative that would avoid direct atmospheric contamination by potentially toxic emissions. The dramatic effects of sludges improving poor soils are encouraging and can be seen in a number of states in the U.S., Illinois and Pennsylvania to mention two. It appears that soils low in organic matter are quickest to respond to such application. Vegetative responses are favorable, odor problems are minimal and atmospheric contamination is significantly reduced. This practice is not, however, without its problems. Sludge application rates to soils low in organic matter can, in some cases, result in mobilization of heavy metals to the underlying aquifer if the soil is unable to stabilize these materials. If organic matter is unable to stabilize nitrogen, nitrates and nitrites may

penetrate the subsoil. Application rates that are too high can result in excess organic accumulation and the overlaying vegetative communities may be adversely affected. The correct application rates depend on many factors, most of which are not understood well at this time.

From a global ecological standpoint, land application of sludge is the best of the two alternatives. Here, the "waste" is not diluted and wasted, but maintained in a relatively concentrated form and to some extent utilized (recycled). Complete recycling will probably not be the case unless all materials are utilized and rates of materials application equal rates of materials removal (steady state). One can also view land disposal of sludges (and wastewater) from the standpoint of putting a future resource (nutrients) in a place where it can be recovered in the future if necessary. By diluting into the atmosphere or in the seas makes it extremely difficult to re-concentrate if needed. Should a land disposal system ultimately "fail" because of over loading it should not be considered an absolute failure but rather an early warning of mismanagement. Fortunately, this mismanagement has been compressed in space, being confined to a small system, that in relative terms would be easier to correct than mismanagement of the atmosphere, for example. By forcing waste management practices into terrestrial systems we also compress potential mismanagement in time. Whereas contamination of the oceans beyond tolerable limits may take hundreds of years, the effects of terrestrial mismanagement may be realized within a single generation. It appears ecologically wise to attack the real problem that of ecological mismanagement, rather than to pass on a more difficult problem to future generations by diluting waste products for there is no assurance that a larger problem can be solved.

#### *Energy Demands:*

Of the proposed alternatives, land disposal will require, by far, the greatest amount of electrical energy over an annual period.



But in terms of total energy utilization this alternative may be equal to or less than the others. If one considers the total amount of energy required to recycle phosphorus, for example, one must consider the energy in returning phosphorus back to productive fields. The land disposal systems have already accomplished this end, and to some extent, the other alternatives will too if sludges are applied to productive land. The chemicals used to maintain the physical-chemical treatment plants are not free in terms of energy. They must be mined, processed and transported to the plant, all of which require energy. In the land disposal alternative it appears to be extremely inefficient to pump raw sewage many miles when the bulk of this material is water. This practice may be desirable in dry climates but the proposed irrigation sites for Michigan are not generally considered water limited. There are no doubt engineering and immediate economic reasons for pumping water but should energetically efficient means for dewatering and transporting sewage be developed; great energy savings may be realized. This is a very important research area and warrants further investigation.

In short, there exists no readily available data base from which to assess the true energy requirements of each alternative. Such an assessment would require a vast systems analysis of our entire U.S. materials flow and is outside of the scope of this study.

#### *Terrestrial Impacts:*

The primary terrestrial impact of the proposed alternatives will be on the land disposal sites. These impacts are described and summarized in detail in Appendix B of this report, "Impact of Wastewater on Land." In short, these impacts are as follows:

1. At wastewater application rates of 2" per week for 35 wks/yr the phosphorus adsorbing function of the soils will be maintained for approximately 30 years.
2. Water leaving the tile drainage system will be of high quality, approximately 0.05 mg/l.

3. Salt loading of the soil does not appear to be a major problem early in the irrigation program but exchangeable sodium will eventually increase to a point that may borderline on dispersion problems in the soil.

4. Potassium and chloride will probably not have adverse affects on the performance of the system.

5. Inactivation of many heavy metals and stabilization of nitrogen will probably occur under the proposed irrigation-cropping regime.

6. The relatively high clay content of the proposed soils should provide a high degree of absorbing capacity for most trace metals.

7. With the exception of boron, possibly arsenic and perhaps selenium there should be little effect on plant uptake and plant growth.

8. Drainage water should meet drinking water standards with the exception of boron and possibly arsenic, iron and selenium.

9. The greatest hazard in the proposed land disposal system appears to be hydraulic overloading. There is reason to believe that an application rate of 70 inches per year would hydraulically overload the proposed soils.

10. Reevaluate the necessity of impoundments, and channelization in tributary streams.

#### *Pesticides and Heavy Metals:*

Of the proposed alternatives examined, land disposal appears to be most effective in removal of pesticides, synthetic organics and heavy metals. The impacts of these on land is discussed in Appendix B. Release of these hazardous materials to water appears greatest in the State of Michigan Plan.

## SECTION VII - SUMMARY AND RECOMMENDATIONS

By way of summarizing consequences of the proposed waste management alternatives we have elected to evaluate the three basic processes rather than the individual alternatives. Since many impacts cannot be quantified we have presented a qualitative summarization in the form of a table (See Table 46).

It is important for the reader to realize that the ecological assessments were based on existing data and that no new field sampling was initiated with the purpose of testing a predetermined hypothesis. Quantitative comparative data on environmental impacts of alternative waste treatment-recycling processes can only come from site specific investigations on the actual systems or scaled down systems themselves. Based on our experience with field oriented ecological impact studies an undertaking of this magnitude would probably take at least 5 years to complete and would require a research budget of at least an order of magnitude above the total budget for the entire Survey Scope study for the Detroit District of the Corps. This figure, however, would represent a very small percentage of the total amount required to implement a single proposed waste management alternative. Below is a list of some of the more important studies that would have to be completed in such an investigation.

1. Research into pretreatment processes to remove heavy metals, pesticides and synthetic organics from industrial wastes.
2. A comprehensive research program designed to characterize the dynamics of hazardous materials in southeastern Michigan.
3. Methods to more completely recycle sludge should be investigated.
4. Efficient ways to dewater raw sewage should be studied.
5. A large scale program to assess the true energy cost of alternative methods of wastewater treatment and recycling should be launched.
6. The feasibility of coupling intensely managed aquatic systems with terrestrial systems for wastewater recycling should be studied.

7. More detailed studies of the nutrient mass balance for Lake Erie and its watershed must be initiated.

8. The feasibility of linking power production systems with waste recycling systems should be investigated.

9. The accumulation of hazardous synthetic organic chemicals and heavy metals into wildlife coming into contact with the large storage lagoons should be fully assessed.

Table 46. Summary of environmental impacts<sup>1</sup> for the three basic waste management strategies.

Impact Zone	Environmental Feature <sup>2</sup>	Management Strategy <sup>3</sup>			
		L.D.	A.W.T.	P.-C.	
GLOBAL	Biogeochemical Cycling	++	+	+	
	Energy Conservation	?	?	?	
	Resource Conservation	++	+	+	
Land Disposal Sites and Adjacent Areas	Soil Quality	+	0	0	
	Hydrology	-	0	0	
	Ecological Stability	-	0	0	
	Productivity	+	0	0	
	Recreation	+	0	0	
	Toxic Materials	-	0	0	
LOCAL	S.E. Mich Tributary Streams	Water Quality	++	++	++
		Hydrology	0	-	-
		Ecological Stability	++	++	++
		Productivity	++	+	+
		Recreation	++	++	++
		Toxic Materials	++	+	+
Lake Erie System	Water Quality	+	+	+	
	Hydrology	0	0	0	
	Ecological Stability	+	+	+	
	Productivity	+	+	+	
	Recreation	++	++	++	
	Toxic Materials	++	+	+	

- <sup>1</sup> ++ Marked improvement over existing conditions  
 + Some improvement over existing conditions  
 0 Little or no change expected  
 - Possible degradation over existing conditions  
 -- Marked degradation over existing conditions

<sup>2</sup> Environmental Feature: See text on following pages for a more complete description of each feature and its relevance to the ecosystem.

<sup>3</sup> Management Strategy: L.D., Land disposal with cropping  
 A.W.T., Advanced biological treatment with phosphorus precipitation  
 P.-C., Physical-chemical treatment



Table 46 (con't.) Explanation of Environmental Features.

1. Biogeochemical Cycling: Approximately one third of the naturally occurring elements in the biosphere are required to support life. Six (C, H, N, O, P, S) are required in relatively large quantities. The continued availability of each of these elements to biota can be upset when essential features of natural recycling mechanisms are disturbed by man.

*An improvement in existing biogeochemical cycling implies that the management strategy would change the rate at which various elements are translocated from compartment to compartment in a direction approaching that of a natural system. Degradation of biogeochemical cycling implies that certain cycling pathways are upset in the direction of "flooding" one portion of the biosphere with too much of an element while providing another portion with too little of the same element.*

2. Energy Conservation: Our current level of technology and food production is highly dependent on sustained inputs of non-solar energy subsidies. Until major breakthroughs are made in the area of power production it is essential to conserve our potentially limited stores of fuel.

*An improvement in energy conservation implies that the proposed alternative is energetically more efficient in cleaning wastewater and recycling constituents in the wastewater than current management technologies aimed toward these objectives. Degradation of energy conservation implies that the proposed alternative is less efficient in cleaning wastewater and recycling materials than current technologies aimed toward these objectives.*

3. Resource Conservation: Certain elements in biogeochemical cycles are needed in large amounts but are relatively scarce in the biosphere. Their continued availability to biota is almost totally dependent on the rate at which the element is biochemically

cycled because geological features of the cycle are too slow in returning the element back to where it can be used again. Phosphorus is an excellent example of such an element. Here, available stores are limited, it is required by all living things and geological recycling processes are too slow to meet biological demands. Current losses to unavailable storage are proceeding at alarming rates.

*An improvement in resource conservation implies that the management strategy will reduce the rate at which non-renewable resources are currently being depleted. A degradation of resource conservation implies that current rates of depletion will be increased.*

4. Soil Quality: The suitability of soil to produce a predictable amount of food is dependent on many physical, chemical and biological factors. Soil enrichment with nutrients is desirable in terms of food production if the concentration of these materials do not exceed toxic levels. It is also desirable if the enrichment process is not accompanied by introduction of undesirable elements (certain heavy metals). Excessive hydraulic loading will also damage soil.

*An improvement over existing conditions implies that the soil, in agricultural terms, will support a new and increased level of plant productivity. This improvement in ecological terms, however, may, in fact, not represent an improvement but rather a short term productivity gain that would ultimately disappear in the long run due to a build up of toxic materials or many other factors characteristic of intensely managed biological systems. A degradation of soil quality implies that the soil in or adjacent to the application site would be rapidly degraded for agricultural purposes.*

5. Water Quality: An improvement in this feature implies that nutrient elements in the aquatic system will be reduced to the point of limiting plant growth. The rate of eutrophication would

*be significantly reduced. Degradation implies the opposite.*

6. Hydrology: Hydrological features of the Southeastern Michigan tributary streams, the Lake Erie systems, and ground water hydrology in and around the land disposal sites may have socioeconomic impacts (shipping, flooding, power generation) and ecological impacts (changes in stream fauna due to changing flow regimes and temperature regimes, water movement between the land, streams, and the aquifer that would be affecting aquatic and terrestrial fauna).

*An improved hydrologic feature for purposes of this report implies changing a previously disrupted hydrologic feature to one that would favor a more stable condition. It does not imply an improved socioeconomic condition. A degraded hydrological feature would imply that diversity and stability of the affected ecosystem were reduced.*

7. Ecological Stability: Stable ecosystems with their complement of diverse functional units are characterized by low fluctuations in the distribution and abundance of organisms and low fluctuations in productivity if the system is insulted by an "outside" stressing agent. The predictability of the stable system to continue its function through time is high. Low stability implies high fluctuations in the distribution and abundance of various species and is characterized by low predictability.

*An improvement in stability would imply that current indices of ecosystem stability in the impact zone would improve. Degradation implies the same in the opposite direction.*

8. Productivity: Increased productivity of those crops used by man can be considered desirable. On the other hand, increased productivity in aquatic ecosystems is often undesirable for both socioeconomic and ecological reasons.

*An improved productivity feature for the land disposal sites implies increased plant productivity that can be used by man.*

*Degradation implies reduced productivity.*

*For the Southeast Michigan tributary streams and the Lake Erie System an improvement in productivity implies a reduction in plant productivity in the direction of reducing the frequency and intensity of algae blooms and reducing the magnitude of diurnal oxygen fluctuations.*

9. Recreation: The current status of portions of the Southeast Michigan tributary streams and Western Lake Erie is quite poor in terms of recreational value. Sport fishing is poor and many other uses are hampered by the unaesthetic appearance of the area. Hunting, fishing, sightseeing and many other types of recreation could potentially improve on improving water quality. Improvement of proposed spray irrigation areas in terms of recreation is very possible if managed wisely.

*An improvement over existing recreational features implies that ecological changes would enhance recreational potential. Degradation implies that the ecological changes induced by an alternative would reduce the recreational potential.*

10. Toxic Materials: Toxic materials (pesticides, heavy metals, PCB's, etc.) introduced into aquatic and terrestrial food chains are hazardous not only to biota of these natural systems but to man as well.

*An improvement in this feature implies that reductions in the total amount of potentially hazardous toxic materials will occur on implementing an alternative. Degradation implies an increase.*



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APPENDIX - A

HEAVY METAL DATA - SOUTHEASTERN MICHIGAN

Table A1. Spectrographic analyses of the heavy metal concentration in Detroit River water - 1962.(1)

ELEMENT	CONCENTRATION (ppb)
Aluminum	410
Barium	100
Beryllium	ND
Cobalt	ND
Chromium	1.5
Copper	43
Iron	250
Lead	15
Manganese	18
Molybdenum	1.6
Nickel	11
Silver	0.21
Tin	ND
Titanium	7.5
Vanadium	ND
Zinc	210

ND = Element analyzed for but was not found.

1. Dufor, C. N. and Becker, E. "Public Water Supplies of the 100 Largest Cities in the United States, 1962" Geological Survey Water-Supply Paper 1812, U.S. Govt. Printing Office, 1964.

Table A2. Heavy metals in large rivers of North America. (1)

Element	Median Value µg/l	Range µg/l	Atlantic Coast	Gulf Coast	Pacific Coast
Silver	0.09	0-0.94	+	0	0
Aluminum	238	12-2550	0	+	0
Cobalt	0	0-5.8			
Chromium	5.8	0.72-84	+	-	-
Copper	5.3	0.83-105	0	+	0
Iron	300	31-1670	0	+	0
Manganese	20	0-185	+	0	0
Molybdenum	0.35	0-6.9	+	0	+
Nickel	10	0-71	+	0	0
Lead	4.0	0-55	0	-	+
Titanium	8.6	0-107	+	+	-
Vanadium	0	0-6.7			
Zinc	0	0-215			

Table (con't.)

key:

- + = significant percentage of determinations above median
- o = approximately 50% of determination above and below median
- = significant percentage of determination below median

1. Durum, W. H. and Haffty, J. "Implications of the Minor Element Content of Some Major Streams of the World," Geochim. Cosmochim. Acta, 27, 1 (1963).



Table A3. Estimated average daily industrial discharge of heavy metals by watersheds (pounds per day).

Total Number of Outfalls	USCOE ID Number	A1	Sb	As	Ba	Cd	Cr	Co	Cu
<u>LAKE ERIE WATERSHED</u>									
1	71-389-1	0	0	0	0	0	0	0	0
2	71-305-1	1506	0	1.5	0	0.3	0.9	0	6.5
3	71-306-1	0	0	0	0	0	0.39	0	0.26
4	71-506-1	14.6	0	0.15	0	0.23	3.0	0	0.8
5	71-507-1	0	0	0	0	0	0	0	0
6	71-734-1	0	0	0	0	0	0.017	0	0
7	71-535-1	0	0	0	0	0	0	0	0.15
TOTALS		1520.6	0	1.65	0	0.53	4.307	0	7.71
<u>MAUMEE RIVER WATERSHED</u>									
1	71-562-1	0	0	0	0	0	0	0	0
2	71-516-1	0	0	0	0	0	0	0	0
3	71-546-1	0	588.9	0	0	0	0	0	85.3
4	71-546-2	0	307	0	0	0	0	0	460
TOTALS		0	895.9	0	0	0	0	0	545.3

Table A3(con't.)

Total Number of Outfalls	USCOE ID Number	Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>LAKE ERIE WATERSHED</u>										
1	71-389-1	1.3	0	0.09	0	0	0	0	0	0.21
2	71-305-1	137	1.5	0	0	0	18.0	0	0	12.9
3	71-306-1	0	0	0	0	0	1.7	0	0	0.21
4	71-506-1	29.0	1.5	0	0	0	2.1	0	0	13.8
5	71-507-1	0	0	0	0	0	0	0	0	0
6	71-734-1	0	0	0	0	0	0	0	0	0.007
7	71-535-1	0.23	0	0	0	0	0	0	0	0
TOTALS		167.5	3.0	0.09	0	0	21.8	0	0	27.13
<u>MAUMEE RIVER WATERSHED</u>										
1	71-562-1	0	0	0	0	0	0	0	0	0.03
2	71-516-1	1580	250	0	0	0	0	0	0	0
3	71-546-1	196	4.9	0	*	0	0	0	0	26.5
4	71-546-2	706	18.4	0	*	0	6.1	36.8	0	184
TOTALS		2482	273.3	0	0	0	6.1	36.8	0	210.53

Table A3 (con't.)

	A1	Sb	As	Ba	Cd	Cr	Co	Cu
<u>RAISIN RIVER WATERSHED</u>								
1	71-582-1	127	0	0	0	0	0	1.2
2	71-582-2	0	0	0	0	0	0	0
3	71-582-9	0	0	0	0	0	0	0
4	71-582-14	0	0	0	0	0	0	0
5	71-579-1	0	0	0	0	0	0	0
6	71-761-1	1.0	0	0	0.05	0.25	0	0.25
7	71-761-2	0.84	0	0	0.17	0.08	0	0.084
8	71-761-3	2.34	0	0	0.117	1.75	0	0.584
9	71-761-4	1208	0	0	0	60.4	0	30.2
10	71-762-1	102	0	0	0	0	0	0.23
11	71-760-1	46	0	0	0	0.046	0	0.07
12	71-1256-1	0	0	0	0	0	0	0.15
13	71-1174-1	0	0	0	0	0	0	0
14	71-329-1	3.52	0	0	0	0	0	5.5
15	71-583-1	0.43	0	0	0	0	0	0.03
16	72-0067-1	0	0	0	0	0.74	0	0.74
17	71-584-1	0	0	0	0	6.006	0	0.004
18	71-584-2	0	0	0	0	0.06	0	0.16
19	71-584-3	0	0	0	0	0.09	0	0.16
20	71-584-4	0	0	0	0	0.002	0	6.65
21	71-584-5	0	0	0	0	0.029	0	0.007
TOTALS		1491	0	0	0.337	62.46	0	40.02

Table A3 (con't.)

		Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>RAISIN RIVER WATERSHED</u>										
1	71-582-1	20	4.7	0	0	0	0	0	2.4	11
2	71-582-2	0	0.09	0	0.009	0	0	0	0	0
3	71-582-9	3.3	0.15	0	0	0	0	0	0	0
4	71-582-14	3.0	0.4	0	0	0	0	0	0	0
5	71-579-1	3.8	0	0	0	0	0	0	0	0
6	71-761-1	0.22	0	0.25	0	0	0.25	0	0	0.50
7	71-761-2	0	0	0.084	0	0	0.084	0	0	0.25
8	71-761-3	12.38	0	0.584	0	0	0.467	0	0	0.98
9	71-761-4	1489	0	60.4	0	0	60.4	0	0	242
10	71-762-1	0	0	0	0	0	0	0	0	2.02
11	71-760-1	0	8.0	0	0	0	0.44	0	0	0.18
12	71-1256-1	0.56	0	0	0	0	0	0	0	0.92
13	71-1174-1	0	0	0	0	0	0	0	0	0
14	71-329-1	22	0	0	0	0	0	0	14.7	5.5
15	71-583-1	0.016	0	0	0.0001	0	0.011	0	0	10.61
16	72-0067-1	0	0.89	0	0	0	0.74	0	0	2.5
17	71-584-1	0	0.006	0	0.005	0	0.058	0	0	0.061
18	71-584-2	0	0.26	0	0.004	0	0.26	0	0	1.7
19	71-584-3	0	0.26	0	0.004	0	0.26	0	0	1.7
20	71-584-4	0	0.86	0	0.02	0	2.2	0	0	5.8
21	71-584-5	0	0.046	0	0.0016	0	0.11	0	0	1.5
TOTALS		1555	15.66	61.38	0.0437	0	65.28	0	17.1	287.22

Table A3 (con't.)

	Al	Sb	As	Ba	Cd	Cr	Co	Cu
<u>HURON RIVER WATERSHED</u>								
1	71-767-1	0	0	0	0	0.01	0	0
2	71-767-2	1.0	0	0	1.0	0	0	0.04
3	71-767-3	0	0	0	0	0	0	0
4	71-767-4	3.0	0	0	3.0	0	0	0
5	71-767-5	0	0	0	0	0.4	0	0
6	71-765-1	2.35	0	0	0	0.14	0	0
7	71-587-1	0	0.004	0	3.7	0.035	0	0
8	71-587-3	0	0.001	0	0.057	0.035	0	0
9	71-587-4	0	0.0018	0	0.115	0.009	0	0
10	71-764-1	0	0	0	0	0.70	0	0
11	71-764-2	0	0	0	0	0.002	0	0
12	71-764-3	0	0	0	0	0.715	0	0
13	71-1097-1	0	0	0	0.009	0.00056	0	0.0079
14	71-1110-1	0	0	0	0	0	0	0
15	71-768-1	0	0	0	0	0.046	0	0
16	71-331-1	0	0	0	0.273	0.149	0	0.198
17	71-522-1	0.117	0	0	0	0	0	0.002
18	71-450-1	2.0	0	0	0	0	0	0
19	71-1014-1	0	0	0	0.0004	0.01	0	0
TOTALS		8.47	0.0068	0	8.16	2.26	0	0.248



Table A3 (con't.)

	Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>HURON RIVER WATERSHED</u>									
1	0	0	0	0	0	0.01	0	0	0
2	2.0	0.1	0	0.004	0	0	0	0	1.0
3	0	0	0	0	0	0.3	0	0	0
4	2.0	0	0	0.0076	0	0	0	0	2.0
5	2.2	0.03	0.03	0	0.02	0.3	0	0	1.0
6	0.09	0	0	0	0	0	0	0	0.26
7	2.7	0	0	0.022	0	0	0	0	0
8	0.1	0	0	0.003	0	0	0	0	0
9	0.23	0	0	0.0057	0	0	0	0	0
10	2.81	0.14	2.82	0	0	0	0	0	0.70
11	0.037	0	0	0	0	0	0	0	0.002
12	5.7	0	0.36	0	0	0	0	0	1.07
13	0.0155	0	0	0	0	0.005	0.013	0	0.0135
14	0.062	0	0	0	0	0	0	0	0
15	0	0	0	0	0	0	0	0	0.154
16	1.496	0	0	0	0	0	0.372	0	4.075
17	0.002	0.001	0	0	0	0	0	0	0
18	2.0	0	0	0	0	0.30	0	0	0.12
19	0	0	0	0	0	0.001	0	0	0
TOTALS	21.45	0.271	3.21	0.0423	0.02	0.916	0.385	0	6.40

Table A3 (con't.)

	Al	Sb	As	Ba	Cd	Cr	Co	Cu
<u>DETROIT RIVER WATERSHED</u>								
1	71-616-1	32.59	0	1664	0	0	0	0
2	71-616-2	33.7	0	1720	0	0	0	0
3	71-618-1	0	0	0	0	27.8	0	33.0
4	71-618-2	0	0	0	0	0.0276	0	0.0985
5	71-618-3	0	0	0	0	2.5	0	5.25
6	71-622-(1-5)	0	0	0	0	0	0	0
7	71-402-1	2.6	0	0	0	0	0	1200
8	71-927-1	0	0	0	0	0.706	0	0
9	71-620-1	0	0	0	0	6.6	0	0
10	71-620-2	0	0	0	0	5.25	0	0
11	71-620-4	0	0	0	0	9.98	0	0
12	71-615-1	0	0	0	0	0.39	0	1.30
13	71-615-2	0	0	0	0	0.20	0	0.83
14	71-334-1	0	0.88	0	0	0	0	0
15	71-334-2	0	0.114	0	0	0	0	0
16	71-333-1	0.9	0	0	0.16	2.3	0.4	7.5
17	71-333-2	0.47	0	0	0.09	0.94	0.23	2.2
18	71-333-3	2.0	0.23	0	0.12	1.2	0.23	4.5
19	71-333-4	1.5	0.16	0	0.05	0.61	0	3.4
20	71-333-5	22	3.4	0	1.4	14.0	0	89.0
21	71-332-1	0	0	0	0	13.6	0	29.94

Table A3 (con't.)

	Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>DETROIT RIVER WATERSHED</u>									
1	71-616-1	2656.5	0	0.0354	0	0	0.354	0	0
2	71-616-2	2.74	0	0.037	0	0	0.366	0	0
3	71-618-1	0	0	0	0	1.12	0	0	35.0
4	71-618-2	0	0	0	0	0.0339	0	0	0.288
5	71-618-3	0	0	0	0	0.72	0	0	3.75
6	71-622-(1-5)	0	0	0	0	0	0	0	0.0006
7	71-402-1	11	0.75	0	0	0	0	0	0.21
8	71-927-1	27.13	0.520	0	0	0.669	0	0	3.34
9	71-620-1	148.6	5.6	0	0	6.6	0	0	42.9
10	71-620-2	118.3	4.73	0	0	5.25	0	0	10.51
11	71-620-4	246.6	5.87	0	0	9.94	0	0	0
12	71-615-1	45.5	0	0	0	0	0	0	3.25
13	71-615-2	3.27	0	0	0	0	0	0	0.83
14	71-334-1	0	0	1.259	0	0	0	0	0
15	71-334-2	0	0	0.06	0	0	0	0	0
16	71-333-1	38	0.56	0.04	0	0	0	0	1.0
17	71-333-2	33.6	0.31	0.02	0	0	0	0	0.59
18	71-333-3	30.0	0.47	0.02	0	0	0	0	0.59
19	71-333-4	20.0	0.19	0.016	0	0	0	3.2	0.4
20	71-333-5	213	4.1	0.34	0	0	0	68.0	8.0
21	71-332-1	5.44	0	0	0	2.72	0	0	23.14

Table A3 (con't.)

	Al	Sb	As	Ba	Cd	Cr	Co	Cu
<u>DETROIT RIVER WATERSHED (con't.)</u>								
22	71-335-1	0	0	0	0.0012	0.008	0.004	0.098
23	71-335-2	0	0	0	0.15	1.12	0.37	4.5
24	71-335-3	0	2.41	0	0.96	26	0	54
25	71-335-4	0	0.04	0	0.017	0.17	0	0.056
26	71-335-5	0	0	0	0.20	1.7	0.52	10.0
27	71-335-6	0	0	0	0.03	0.22	0.08	1.1
28	71-614-1	0	0	0	0	0.001	0	0.002
29	71-612-1	0	1.54	0	0.0	0.0	0	0.0
30	71-612-2	0	-0.023	0	0.0	0.0	0	0.0
31	71-612-3	0	0.366	0	0.0	0.0	0	0.0
32	71-612-4	0	0.760	0	0.0	0.0	0	0.0
33	71-612-5	0	0.088	0	0.0	0.0	0	0.0
34	71-612-6	0	4.06	0	0.0	0.0	0	0.0
35	71-612-7	0	0.092	0	0.0	0.0	0	0.0
36	71-612-8	0	-0.40	0	0.0	0.0	0	0.0
37	71-612-9	0	0.0	0	0.0	0.0	0	0.0
38	71-612-10	0	0.003	0	0.0	0.0	0	0.0
39	71-612-11	0	0.0	0	0.0	0.0	0	0.0
40	71-612-12	0	0.135	0	0.0	0.0	0	0.0
41	71-612-13	0	0.0	0	0.0	0.214	0	0.0
42	71-612-14	0	1.35	0	0.0	0.675	0	0.0
43	71-612-15	0	5.07	0	0.0	21.7	0	36.3
44	71-612-16	0	0.048	0	0.0	0.0	0	0.0

Table A3 (con't.)

	Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>DETROIT RIVER WATERSHED (con't.)</u>									
22	71-335-1	0.50	0.006	0.0017	0.0004	0	0	0	0.01
23	71-335-2	34	0.41	0.15	0.04	0	0	0	0.94
24	71-335-3	310	17	0	0.24	0	0	48	6.0
25	71-335-4	2.5	0.075	0	0.004	0	0	0.83	0.10
26	71-335-5	33.0	0.63	0.21	0.05	0	0	0	1.3
27	71-335-6	7.7	0.17	0.03	0.008	0	0	0	0.21
28	71-614-1	0	0.012	0	0	0.006	0	0	0.007
29	71-612-1	18.6	-0.64	0	0.0	0.0	0.0	0.705	1.28
30	71-612-2	30.2	-0.24	0	0.071	0.0	0.0	0	1.65
31	71-612-3	30.2	0.0	0	0.0	0.0	0.0	0	0.0
32	71-612-4	27.0	0.0	0	0.0	0.760	0.0	0	9.33
33	71-612-5	8.0	-0.063	0.769	0.0	0.069	0.0	0	0.0
34	71-612-6	2330	11.8	8.96	1.35	1.86	0.0	0	140
35	71-612-7	1.75	0.0	0	0.0	0.0	0.0	0	0.605
36	71-612-8	194	-2.00	10.6	0.0	0.0	0.0	0	35
37	71-612-9	1450	23.5	0	0.0	0.0	0.0	0	0.0
38	71-612-10	-0.133	0.008	0	0.0	0.004	0.0	0	0.029
39	71-612-11	174	1.11	0	0.0	0.554	0.0	0	0.0
40	71-612-12	0.225	0.0	0	0.0	0.038	0.0	0	0.0
41	71-612-13	4.26	0.0	0	0.0	0.378	0.0	0	0.0
42	71-612-14	32.4	1.35	0	0.0	0.0	0.0	0	0.675
43	71-612-15	12,200	7.25	0	2.17	21.7	0.0	0	48.9
44	71-612-16	9.98	0.0	0	0.143	0.238	0.0	0	-0.238



Table A3 (con't.)

	Al	Sb	As	Ba	Cd	Cr	Co	Cu
<u>DETROIT RIVER WATERSHED (con't.)</u>								
45	71-612-17	0	3.04	0	0.0	0.0	0	0.0
46	71-612-18	0	0.523	0	0.0	0.0	0	0.0
47	71-612-19	0	0.0	0	0.0	0.083	0	0.0
TOTALS	126.28	0	23.89	3384	3.18	138.00	1.83	1446.78
<u>ROUGE RIVER WATERSHED</u>								
1	71-337-1	0	0	0	0	1.0	0	0
2	72-0050-1	0	0	0	0	0.0002	0	0
3	72-0014-1	0	0	0	0	7.5	0	0
4	71-777-1	0	0	0	0	0.093	0	0
5	71-779-1	0	0	0	0	0.13	0	0
6	71-778-1	0	0	0	0	0.003	0	0
7	71-987-1	0	0	0	0	0	0	0
8	71-610-1	2.3	0	0	0	0	0	0
9	71-403-1	0	0	0	0	0	0	0
10	71-492-1	0	0	0	0	0	0	0
11	71-405-1	0	0	0	0	5.75	0	0.85
12	71-406-1	0.51	0.05	0	0.026	0.128	0	0.128
13	71-404-1	0.0143	0	0	0	0.00034	0	0
14	71-776-1	0	0	0	0	0	0	0
15	71-776-2	0	0	0	0	0	0	0
16	71-776-3	295	0	0	6.7	0	0	22

Table A3(con't.)

	Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>DETROIT RIVER WATERSHED (con't.)</u>									
45	71-612-17	146	0	2.85	0	0.0	0.0	0	0.0
46	71-612-18	14.4	0	0.142	0	0.0	0.0	0	0.95
47	71-612-19	-0.167	0	0.025	0	0.042	0.0	0	0.0
TOTALS		20,658.10	83.48	23.62	8.8484	2.69	50.01	0.72	120.74 380.85
<u>ROUGE RIVER WATERSHED</u>									
1	71-337-1	0	0	0	0	0	0	0	1.0
2	72-0050-1	0	0	0	0	0	0	0	0
3	72-0014-1	0	0	0	0	0	0	0	1.5
4	71-777-1	0	0.0117	0	0	0	0	0	0.28
5	71-779-1	0	0	0	0	0	0	0	0.01
6	71-778-1	0.092	0.833	0	0	0	0	0	0.007
7	71-987-1	0	0	0	0	0	0	0	0
8	71-610-1	0.5	0	0	0	0.22	0.022	0	0.022
9	71-403-1	0	0	0	0	0.21	0	0.382	0.44
10	71-492-1	0	0	0	0	0	0	0	0.023
11	71-405-1	3.73	0.54	0	0	8.0	0	0	0.85
12	71-406-1	0.25	0.26	0.002	0	0.128	0.26	0	0.22
13	71-404-1	0.093	0	0	0	0	0	0	0
14	71-776-1	469	0	0	0	0	0	0	0
15	71-775-2	38	0	0	0	0	0	0	5.0
16	71-776-3	862	178	0	0	0	0	0	600

Table A3 (con't.)

	Al	Sb	As	Ba	Cd	Cr	Co	Cu
<u>ROUGE RIVER WATERSHED (con't.)</u>								
17	71-776-4	375	0	0	0	0	0	0
18	71-776-5	89	0	0	2.0	0	0	0
19	71-776-6	0	0	0	0	0	0	0
20	71-866-1	408	1.3	0	0.0	0.23	0	0
21	71-866-2	230	2.66	0	0.0	0.16	0	0
TOTALS		1399.82	4.01	0	8.73	14.99	0	22.98
<u>CLINTON RIVER WATERSHED</u>								
1	71-407-1	0	0	0	0	0	0	0
2	71-410-1	0	0	0	0	0.015	0	0
3	71-409-1	0	0	0	0	0	0	0
4	71-409-2	0	0	0	0	0	0	0
5	71-490-1	0	0	0	0	0	0	0
6	71-339-1	0	0	0	0	0	0	0
7	71-270-1	0.502	0	0	0	1.004	0	0
8	71-488-1	0	0	0	0	0	0	0.001
9	71-783-1	0.004	0	0	0	0	0	0.004
10	71-784-1	0	0	0	0	0.042	0	0
11	71-781-1	0	0	0	0	0	0	0
12	71-341-1	0	0	0	0	0	0	0

Table A3 (con't.)

	Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>ROUGE RIVER WATERSHED (con't.)</u>									
17	71-776-4	1250	0	188	0	0	0	0	525
18	71-776-5	425	72	126	0	0	5.5	0	456
19	71-776-6	1200	0	0	0	0	0	0	427
20	71-866-1	198	0.25	0	0.0	0	0	0.66	2.0
21	71-866-2	67	0.37	0	0.0	0	0	1.6	2.8
TOTALS		4513.66	288.72	653.64	0.003	0	8.56	2.642	2022.15
<u>CLINTON RIVER WATERSHED</u>									
1	71-407-1	650	20	5.0	0	0	0	0	0
2	71-410-1	0.016	0	0	0	0	0	0.062	0.125
3	71-409-1	120	0	0	0	0	0	0	0.239
4	71-409-2	0.467	0	0	0	0	0	0	0
5	71-490-1	0	0	0	0	0	0	0	0
6	71-339-1	0	0	0	0	0	0	0	0
7	71-270-1	2.46	0	0.05	0	0	0	0	0
8	71-488-1	0	0	0.005	0	0	0	0	0
9	71-783-1	0.026	0	0.007	0.001	0.001	0.001	0	0
10	71-784-1	0.11	0	0	0	0	0	0	0
11	71-781-1	0	0	0.5	0	0	0	0	0
12	71-341-1	0	0	0	0	0	0	0	0.042

Table A3(con't.)

	Al	Sb	As	Ba	Cd	Cr	Co	Cu
<u>CLINTON RIVER WATERSHED (con't.)</u>								
13	71-640-1	0	0	0	0	0.13	0	0.013
14	71-1259-1	0	0.033	0	0.033	0.166	0	0.166
TOTALS		0.506	0.033	0	0.033	1.356	0	0.184
<u>ST. CLAIR RIVER WATERSHED</u>								
1	71-411-2	0	0	0	0	0.025	0	0
2	71-785-1	0	0	0	0	2.5	0	0.25
3	71-487-1	0.02	0	0	0	0	0	0
4	71-495-1	0	0	0	0	0	0	0
5	71-935-1	0	0	0	0	0	0	0
TOTALS		0.02	0	0	0	2.53	0	0.25
<u>PINE RIVER WATERSHED</u>								
1	71-415-1	1.2	0	0	0	0.28	0	0.6
2	71-412-1	0.13	0.0002	0	0.001	0.02	0	0.04
3	71-411-2	2.5	0	0	0.5	2.5	0	2.5
TOTALS		3.83	1.0002	0	0.501	2.80	0	3.14
<u>BLACK RIVER WATERSHED</u>								
1	71-594-2	0	0	0	0	1.23	0	5.06
2	71-594-3	0	0	0	0	0.57	0	3.46
3	71-594-4	0	0	0	0	0.42	0	3.24



Table A3 (con't.)

	Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>CLINTON RIVER WATERSHED (con't.)</u>									
13	71-640-1	0	0.26	0	0.26	0	2.6	0	0.013
14	71-1259-1	3.65	2.98	0	0.0003	0	0.166	0	0.830
TOTALS	776.74	23.24	915.06	0.2603	0.001	2.767	0.167	0.062	1.246
<u>ST. CLAIR RIVER WATERSHED</u>									
1	71-411-2	0.064	0.003	0	0	0.003	0	0	17.21
2	71-785-1	0	0.51	0	0.51	50.8	0	0	0.25
3	71-487-1	0.57	0	0	0	0	0	0	0
4	71-485-1	0	0	0	0	0	0	0	0
5	71-935-1	0	0	0	0	0	0	0	0
TOTALS	0.63	0.51	0.33	0.51	0	50.8	0	0	17.46
<u>PINE RIVER WATERSHED</u>									
1	71-415-1	1.0	0	0	0	1.7	0	0	0
2	71-412-1	17.0	0.003	0	0	1.3	0	0	0.06
3	71-412-2	33	0.5	0.05	0	2.5	0	0	2.5
TOTALS	51.0	0.50	2.5	0.05	0	5.5	0	0	2.56
<u>BLACK RIVER WATERSHED</u>									
1	71-594-2	0	0	0.008	0	0.45	0	100	0.185
2	71-594-3	0	0	0	0	0.36	0	0	0.32
3	71-594-4	0	0	0.0156	0	2.17	0	0	0.222

Table A3 (con't.)

	Al	Sb	As	Ba	Cd	Cr	Co	Cu
<u>BLACK RIVER WATERSHED (con't.)</u>								
4	71-594-5	0	0	0	0	1.12	0	2.45
5	71-594-6	0	0	0	0	0.67	0	3.71
6	71-594-7	0	0	0	0	0.27	0	1.21
7	71-527-	0	0	0	0	0	0	0
TOTALS	0	0	0	0	0	4.28	0	19.13
GRAND TOTAL (DAILY)	4551.	896.	30.59	3384	21.47	23.3	1.83	2086
GRAND TOTAL (YEARLY)*	1,183,172	232,960	7953	879,840	5582	60,580	476	542,360

USCOE = U.S. Army Corps of Engineers, Detroit District, Box 1027, Detroit, Michigan 48231

Table A3 (con't.)

	Fe	Pb	Mn	Hg	Mo	Ni	Sn	Ti	Zn
<u>BLACK RIVER WATERSHED (con't.)</u>									
4	71-594-5	0	0	0.003	0	0.93	0	20	0.10
5	71-594-6	0	0	0.006	0	0.95	0	5	0.134
6	71-594-7	0	0	0.006	0	0.01	0	0	0.20
7	71-527-1	0	0	0	0	0	0	0	0
TOTALS		0	0	0.0386	0	4.87	0	125	1.16
GRAND TOTAL (DAILY)	30,220	689	1660	9.80	2.71	217	43.86	266	2957
GRAND TOTAL (YEARLY)*	7,857,200	179,140	431,600	2519	705	56,420	11,403	69,160	768,820

\*Based on 260 working days per year.

Table A4. Heavy metal analyses of selected municipal wastewater treatment plant influents and effluent.

Element	San Antonio Rilling Plant Average Value for 1966 <sup>1</sup>		Lawrence, Kansas Municipal <sup>2</sup>		Boulder, Colorado White Rocks Plant <sup>3</sup>		Warren Michigan Municipal <sup>4</sup>		Ypsilanti, Michigan Municipal Average Value Jan. - Aug., 72 <sup>4</sup>		Detroit, Michigan Municipal <sup>5</sup>	
	Raw	Final	Raw	Final	Raw	Final	Raw	Final	Raw	Final	Raw	Final
Aluminum	2295	520	-	-	-	-	-	-	-	-	-	-
Arsenic	<100	<100	2.7	1.8	-	-	-	-	-	-	-	-
Barium	98	44	-	-	-	-	-	-	-	-	-	-
Beryllium	<0.1	<0.1	-	-	-	-	-	-	-	-	-	-
Cadmium	<20	<20	-	-	<3.7	-	0	-	100	-	37	19
Chromium	237	54	-	-	<5.5	-	0	-	-	-	800	360
Cobalt	<10	<10	-	-	-	-	-	-	-	-	-	-
Copper	86	34	-	-	-	-	-	100	-	-	530	190
Iron	852	210	-	-	-	-	-	1000	1050	17,800	9,800	-
Lead	55	20	-	-	-	-	-	-	-	-	-	-
Manganese	56	56	-	-	-	-	-	-	-	-	-	-
Mercury	-	-	-	-	-	-	-	-	-	-	-	-
Molybdenum	<20	<20	-	-	-	-	-	-	-	-	-	-
Nickel	<10	<10	-	-	-	-	-	-	-	-	690	410
Selenium	-	-	-	-	<2.3	-	-	-	-	-	-	-

Table A4 (con't.)

Silver	13	11	-	-	-	4.6	-	-	-	-	-	-
Tin	-	-	-	-	-	-	-	-	-	-	-	-
Titanium	-	-	-	-	-	-	-	-	-	-	-	-
Zinc	360	131	-	-	-	-	-	750	-	-	1,520	740



Table A4 (con't.)

Element	Detroit, Mich. Activated Sludge Pilot Plant <sup>5</sup>		S. D. Warren Paper Mill, Muskegon, Mich. Effluent <sup>6</sup>		Muskegon Effluent Composite Average April, 1970 <sup>7</sup>		Muskegon Heights Effluent Composite Average May, 1970 <sup>7</sup>		Muskegon, Michigan Composite Average <sup>7</sup>		Typical Secondary Effluent, Heavy Metal Levels <sup>8</sup>	
	Raw	Final	Raw	Final	Raw	Final	Raw	Final	Raw	Final	Raw	Final
Aluminum	-	-	-	8300	-	70	-	10	-	4900	-	-
Arsenic	-	-	-	-	-	10	-	290	0	20	-	-
Barium	-	-	-	-	-	-	-	-	-	-	-	-
Beryllium	-	-	-	-	-	-	-	-	-	-	-	-
Cadmium	35	19	-	-	-	33	-	15	-	12	-	100
Chromium	770	100	-	35	-	1450	-	100	-	510	-	200
Cobalt	-	-	-	-	-	-	-	-	-	-	-	-
Copper	580	80	-	29	-	-	-	62	-	52	-	100
Iron	22,700	3460	-	41	-	990	-	700	-	610	-	100
Lead	-	-	-	28	-	510	-	300	-	200	-	100
Manganese	-	-	-	-	-	150	-	150	-	60	-	200
Mercury	-	-	-	1.0	-	-	-	-	-	1.0	-	5.0
Molybdenum	-	-	-	-	-	-	-	-	-	-	-	-
Nickel	600	260	-	28	-	380	-	70	-	150	-	200
Selenium	-	-	-	-	-	70	-	290	-	41	-	-

Table A4 (con't.)

Silver	-	-	-	-	-	-	-	-	-	-
Tin	-	-	-	-	100	-	100	-	30	-
Titanium	-	-	-	-	-	-	-	-	-	-
Zinc	1,060	220	-	132	1180	-	140	-	480	200

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5. Shannon, A. M., Detroit Metropolitan Water Department, 735 Randolph St., Detroit, Michigan 48226, Personal Comm., 1972.
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Table A5. Contributions of fuels to lead, cadmium, and copper to the Chicago area suspended particulate.<sup>(1)</sup>

	Coal	Coke	Fuel Oil	Gasoline
Combustion estimate (million tons per year)	20	15	7	8
Particulate emission estimate (thousand tons per year)	220	18	14	*
Lead emission estimate (tons per year)	300	22	30	1800
Cadmium emission estimate (tons per year)	11	1	--	--
Copper emission estimate (tons per year)	100	7	26	--

\*Assumed 2 grams of lead per gallon and 25% of emissions airborne.

(1) - Harrison, P. R. and Winchester, J. W. "Area-Wide Distribution of Lead, Copper, and Cadmium in Air Particulates from Chicago and Northwest Indiana," Atmospheric Environment, 5 863 (1971).

Table A6. Airborne lead concentrations in the San Diego, California area.<sup>(1)</sup>

Sampling Location	Year	CONCENTRATION (ng/m <sup>3</sup> )							Yearly Mean
		Jan - Mar	Apr - Jun	Jul - Sept	Oct - Dec				
Downtown San Diego Commercial area	1967	3500	1160	1040	2950			2160	
	1968	3270	1130	1330	3650			2380	
	1969	2570	1050	1070	4050			2200	
Mission Valley 10 miles inland from ocean - increasing urban development	1967	--	--	1070	1870			--	
	1968	1760	1270	1550	2940			1900	
	1969	2050	1480	1520	3140			2060	
Scripps Pier extends 1000 feet into Pacific Ocean	1967	--	--	270	610			--	
	1968	580	240	250	610			420	
	1969	400	210	230	630			370	
Laguna Mountain Elevation 6100 feet - remote from industrial pollution	1968	--		--	40			--	
	1969	10	0	70	50			50	

(1) - Anon. "Lead Concentrations in City Air Increases." Chemical and Engineering News, 48 (10) 42 (1970).

Table A7. Airborne heavy metal concentrations in ng/m<sup>3</sup> for selected cities in the United States.

Element	Chicago & N.W. Indiana <sup>1</sup>			E. Chicago, Gary Whiting, Hammond & N.W. Indiana <sup>2</sup>			Bronx New York <sup>5</sup>	Lower Manhattan New York <sup>5</sup>	Tuxedo* New York <sup>5</sup>	Manhattan New York <sup>6</sup>
Cadmium	Max	80	-	-	-	-	22	36	5	4
	Mean	19	-	-	-	-	14	23	3	-
	Min	<5	-	-	-	-	6	9	1	0
Lead	Max	7,000	-	-	-	-	5030	4150	620	3000
	Mean	1,900	-	-	-	-	3820	2990	409	1500
	Min	100	-	-	-	-	2540	1750	110	100
Copper	Max	10,000	-	-	-	-	270	300	86	250
	Mean	1,000	-	-	-	-	133	212	44	110
	Min	<20	-	-	-	-	30	90	12	30
Iron	Max	-	-	-	-	-	-	-	-	-
	Mean	-	-	-	-	-	-	-	-	-
	Min	-	-	-	-	-	-	-	-	-
Aluminum	Max	-	-	-	-	-	-	-	-	-
	Mean	-	-	-	-	-	-	-	-	-
	Min	-	-	-	-	-	-	-	-	-
Zinc	Max	-	-	-	-	-	1440	4180	260	1660
	Mean	-	-	-	-	-	790	2380	210	700
	Min	-	-	-	-	-	150	830	60	10
Manganese	Max	-	-	-	-	-	81	97	57	70
	Mean	-	-	-	-	-	54	71	33	40
	Min	-	-	-	-	-	33	35	9	10
Titanium	Max	-	-	-	-	-	-	-	-	-
	Mean	-	-	-	-	-	-	-	-	-
	Min	-	-	-	-	-	-	-	-	-



Table A7 (con't.)

Chromium	Max	-	113±20	93	93	23	35
	Mean	-	33	49	63	9	16
	Min	-	6.2±0.8	17	27	3	0
Antimony	Max	-	31±3	-	-	-	-
	Mean	-	8.95	-	-	-	-
	Min	-	2.2±0.2	-	-	-	-
Arsenic	Max	-	12±2	-	-	-	-
	Mean	-	5.1	-	-	-	-
	Min	-	2±1	-	-	-	-
Silver	Max	-	5±2	-	-	-	-
	Mean	-	<1.7	-	-	-	-
	Min	-	<0.5	-	-	-	-
Mercury	Max	-	4.9±0.9	-	-	-	-
	Mean	-	2.29	-	-	-	-
	Min	-	0.8±0.3	-	-	-	-
Selenium	Max	-	4.4±1.2	-	-	-	-
	Mean	-	2.33	-	-	-	-
	Min	-	0.8±0.5	-	-	-	-
Cobalt	Max	-	2.6±0.6	-	-	-	-
	Mean	-	1.20	-	-	-	-
	Min	-	0.47±0.06	-	-	-	-
Nickel	Max	-	-	250	210	160	250
	Mean	-	-	150	160	68	140
	Min	-	-	50	70	10	50

\*Tuxedo, N. Y. served as background station and is located about 50 km northwest of New York City.

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2. Harrison, P. R., Rahn, K. A., Dams, R., Robbins, J. A. and Winchester, J. W. "Area-Wide Trace Metal Concentrations Measured by Multielement Neutron Activation Analysis - A One Day Study in Northwest Indiana." J. Air Poll. Cont. Assoc., 21, 563 (1971).
5. Kneip, T. J., Eisenbud, M., Strehlow, C. D. and Freudenthal, P. C. "Airborne Particulates in New York City," J. Air Poll. Cont. Assoc., 20, 144 (1970).
6. U.S.P.H.S. "Air Quality Data From the National Air Sampling Networks and Contributing State and Local Networks 1964-1965," U.S. Dept. of Health, Education and Welfare, Cincinnati, Ohio (1966).

Table A8. Heavy metal concentrations in the air of 58 cities and 29 nonurban areas of the National Air Sampling Network.<sup>(1)</sup>

Element	Number of Urban Cities Present	Range ng/m <sup>3</sup>	Number of Nonurban Cities Present	Range ng/m <sup>3</sup>
Chromium	45/58	1 - 55	29/29	1 - 12
Titanium	53/58	10 - 70	29/29	3 - 18
Nickel	56/58	1 - 118	28/29	0.6 - 12
Manganese	58/58	10 - 1440	29/29	5 - 47
Iron	58/58	300 - 4200	29/29	80 - 700
Cobalt	1/58	6	0/29	-
Copper	58/58	10 - 570	29/29	10 - 250
Zinc	43/58	10 - 840	27/29	13 - 200
Molybdenum	15/58	10 - 30	23/29	0.1 - 3.2
Vanadium	44/58	1 - 458	29/29	0.5 - 24
Barium	33/58	2 - 1500	0/29	-
Beryllium	9/58	1 - 2	4/29	~ 0.13
Cadmium	36/58	2 - 370	17/29	0.4 - 26
Tin	30/58	10 - 30	15/29	0.2 - 1.8
Antimony	4/58	42 - 85	3/29	1 - 2
Lead	58/58	100 - 2300	28/29	2 - 150

(1) Schroeder, H. A. "A Sensible Look at Air Pollution by Metals," Arch. Environ. Health, 21 798 (1970).

Table A9. United States Geological Survey reconnaissance of selected minor elements in surface waters of southeastern Michigan (concentration in  $\mu\text{g/l}$  or ppb). (1)

Sampling Location	Source	Date	Arsenic Dissolved	Cadmium Dissolved	Chromium (Hexavalent) Dissolved	Cobalt Dissolved	Lead Dissolved	Zinc Dissolved	Mercury Dissolved	Mercury Total
1. St. Clair River at Port Huron	PWS	10/19/70	<10.0	<1.0	<1.0	<1.0	5	160	<0.5	<0.5
2. Detroit River at Detroit	PWS	10/16/70	<10.0	<1.0	<1.0	<1.0	4	10	<0.5	<0.5
3. River Rouge cut- off Canal at Zug Island	MIC	10/14/70	<10.0	<1.0	<1.0	<1.0	4	90	<0.5	<0.5
4. Detroit River, Trenton Channel at Swan Island	MIC	10/14/70	10	<1.0	<1.0	<1.0	4	30	0.2	0.6
5. Detroit River Livingstons Channel below Amherstburg	MIC	10/14/70	10	1	<1.0	<1.0	2	120	0.2	<0.5
6. Ford Lake, Huron River at Ypsilanti	MIC	10/14/70	<10.0	<1.0	<1.0	<1.0	3	100	0.1	<0.5
7. Lake Erie near Pointe Aux Peaux	PWS	10/14/70	<10.0	1	<1.0	<1.0	3	110	0.1	<0.5
8. River Raisin at mouth near Monroe	MIC	10/14/70	30	1	<1.0	<1.0	4	160	<0.5	<0.5

PWS = Public water supplies for cities of more than 100,000 population.

MIC = Water courses downstream from major municipal and/or industrial complexes.

- (1) - Durum, W. H., Hem, J. D. and Heidel, S. G. "Reconnaissance of Selected Minor Elements in Surface Waters of the United States, October, 1970." Geological Survey Circular 643, U.S. Dept. of Interior, Washington, D. C., 1971, 49 pp.



Table A10. The dissolved heavy metal concentrations in micrograms per liter (ppb) in water samples taken from the Maumee River and its tributaries between July and August, 1971.(1)

Sampling Site Location	Date	Copper	Zinc	Nickel	Cadmium	Chromium (Hexavalent)	Mercury*	Lead	Arsenic
Mill Street Drain at M-49 1/2 mile north of the Michigan - Ohio line	7/30/71	<10	<10	<10	<1	<10	<0.1	<10	<10
West fork of the west branch of the St. Joseph River at M-49 in Camden	7/30/71	<10	<10	<10	<1	<10	<0.1	<10	<10
East fork of the west branch of the St. Joseph River at Territorial Road	7/30/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Silver Creek at Territorial Road	7/30/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Unnamed tributary to the east branch of St. Joseph River at Miner Road downstream from Pittsford	7/29/71	<10	<10	<10	<1	<10	<0.1	<10	<10
East branch of St. Joseph River at Williams County Road 9 in Ohio	7/30/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Myers Drain at Territorial Road	7/30/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Bean Creek in the impoundment in Addison at Comstock Street	7/29/71	<10	<10	<10	<1	<10	<0.1	<10	<10

Table A10 (con't.)

Bean Creek at US-127 south of Addison	7/29/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Garrison Drain at State Street in Hudson	7/29/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Bean Creek at Nelson Road south of Hudson	7/29/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Bean Creek at Main Street in Morenci	7/30/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Bean Creek at Fulton County Road T in Ohio	7/30/71	<10	10	<10	<1	<10	<0.1	<10	<10
Silver Creek at M-156 in Morenci	7/30/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Halfway Creek at Smith Road	8/02/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Halfway Creek at State Line Road in Lucas County, Ohio	8/02/71	<10	<10	<10	<1	<10	<0.1	<10	<10
Halfway Creek at alternate US-24	8/02/71	--	--	--	--	--	<0.1	--	--
Silver Creek 75 yards upstream from its confluence with Halfway Creek	8/02/71	--	--	--	--	--	<0.1	--	--
Shantee Creek at alternate US-24	8/02/71	--	--	--	--	--	<0.1	--	--

Table A10(con't.)

\* Total mercury analysis, -- Chemical parameter not measured.

(1) WRC Interstate Rivers Monitoring Program Metal Analysis, 1963-1971, Michigan Water Resources Commission,  
Stevens T. Mason Bldg., Lansing, Michigan.

Table A11. The dissolved heavy metal concentrations in micrograms per liter (ppb) in water samples taken from the sampling sites noted between 1969 and 1971.<sup>(1)</sup>

Sampling Location	Date	Copper	Zinc	Nickel	Chromium Hexavalent	Chromium Trivalent	Mercury	Lead	Arsenic	Antimony	Cadmium
Clinton River	June 18, 1969	-	0	-	-	-	-	-	-	-	-
Clinton River	February 8, 1971	-	-	-	-	-	-	-	2	-	-
Clinton River	June 22, 1971	<10	5	<10	<10	-	<0.02 <sup>a</sup>	<10	<2	-	<1
Belle River	June 18, 1969	-	0	-	-	-	-	-	-	-	-
Belle River	February 8, 1971	-	-	-	-	-	-	-	2	-	-
Belle River	June 22, 1971	<10	10	<10	<10	-	<0.02 <sup>a</sup>	<10	<2	-	<1
Pine River	June 18, 1969	-	0	-	-	-	-	-	-	-	-
Pine River	February 8, 1971	-	-	-	-	-	-	-	2	-	-
Pine River	June 22, 1971	10	<10	<10	<10	-	<0.02 <sup>a</sup>	<10	<2	-	<1
Black River	June 18, 1969	-	0	-	-	-	-	-	-	-	-
Black River	February 8, 1971	-	-	-	-	-	-	-	<2	-	-
Black River	June 22, 1971	<10	10	<10	<10	-	<0.02 <sup>a</sup>	<10	<2	-	<1

Sampling Site Descriptions:

Clinton River - Clinton River at Bridgeview Road Bridge 3 miles east of Mt. Clemens, Michigan  
 Belle River - Belle River at bridge in Marine City  
 Pine River - Pine River at M-29 Bridge in St. Clair  
 Black River - Black River at end of Catwalk downstream from C&O Railway Bridge at the end of Clyde Street in Port Huron

Table AII(con't.)

a = Value represents the total concentration of heavy metal (dissolved and particulate).  
o = Chemical parameter less than analytically detectable level.  
- = Chemical parameter not measured.

(1) WRC Interstate Rivers Monitoring Program Metal Analysis, 1963-1971, Michigan Water Resources Commission, Stevens T. Mason Bldg., Lansing, Michigan.



Table A12. The dissolved heavy metal concentrations in micrograms per liter (ppb) in water samples taken from the Huron River at Rockwood, Michigan between 1969 and 1971.

Date	Copper	Zinc	Nickel	Chromium Hexavalent	Chromium Trivalent	Mercury	Lead	Arsenic	Antimony	Cadmium
January 24, 1969	20*	20*	20*	0*	-	-	-	-	-	-
February 13, 1969	10*	30*	10*	150*	-	-	-	-	-	-
March 10, 1969	20*	10*	20*	10*	-	-	-	-	-	-
April 9, 1969	0	0	0	0	-	-	0	-	-	0
May 28, 1969	0	10	-	-	-	-	-	-	-	-
June 18, 1969	-	10	-	0	0	-	-	-	-	-
July 7, 1969	0	20	0	0	-	-	-	-	-	0
September 15, 1969	40	10	30	-	-	-	-	-	-	0
October 10, 1969	0	20	0	0	-	-	0	-	-	0
November 17, 1969	0	10	40	0	-	-	0	-	-	0
January 21, 1970	0	10	20	0	-	-	0	-	0	0
February 16, 1970	10	20	10	0	-	-	-	-	-	0
March 16, 1970	30	40	0	0	-	-	-	-	-	0
April 13, 1970	0	13	10	0	-	-	-	-	-	0
May 5, 1970	0	10	0	0	-	-	-	-	-	11

Table A12 (con't.)

June 4, 1970	0	12	10	0	-	-	-	-	0
October 12, 1970	20	0	0	0	-	-	-	-	0
November 9, 1970	0	0	0	0	-	-	-	-	0
February 9, 1971	-	-	-	-	-	-	-	<2	-
June 22, 1971	<10	10	<10	<10	-	<0.02 <sup>a</sup>	<10	5	<1
October 14, 1971	0 <sup>a</sup>	30 <sup>a</sup>	<10 <sup>a</sup>	-	-	<0.02 <sup>a</sup>	<10 <sup>a</sup>	-	<1

Sampling Site Description - Huron River at Dixie Highway 3.5 miles southeast of Rockwood, Michigan

Key

\* = Questionable results

a = Value represents total concentration of heavy metal (dissolved and particulate).

0 = Chemical parameter is less than analytically detectable level.

- = Chemical parameter not measured.

(1) WRC Interstate Rivers Monitoring Program Metal Analysis 1963-1971, Michigan Water Resources Commission, Stevens T. Mason Bldg., Lansing, Michigan.

Table A13. The dissolved heavy metal concentrations in micrograms per liter (ppb) in water samples taken from the Raisin River at Monroe, Michigan between 1969 and 1971.

Date	Copper	Zinc	Nickel	Chromium Hexavalent	Chromium Trivalent	Mercury	Lead	Arsenic	Antimony	Cadmium
January 24, 1969	10*	10*	30*	100*	-	-	-	-	-	-
February 13, 1969	10*	350*	10*	100*	-	-	-	-	-	-
March 10, 1969	20*	20*	20*	10*	-	-	-	-	-	-
April 9, 1969	0	40	0	0	-	-	0	-	-	0
May 28, 1969	0	20	-	-	-	-	-	-	-	-
June 18, 1969	-	30	-	0	0	-	-	-	-	-
July 7, 1969	0	50	0	0	-	-	-	-	-	0
September 15, 1969	20	30	30	-	-	-	-	-	-	0
October 10, 1969	60	80	30	0	-	-	0	-	-	0
November 17, 1969	10	50	40	20	-	-	0	-	-	0
January 21, 1970	70	50	30	100	-	-	0	-	-	0
February 16, 1970	50	40	10	50	-	-	-	-	-	0
March 16, 1970	30	40	0	0	-	-	-	-	-	0
April 13, 1970	30	23	20	30	-	-	-	-	-	0
May 5, 1970	32	14	10	70	-	-	-	-	-	5

Table A13 (con't.)

June 4, 1970	40	23	0	40	-	-	-	-	0
October 12, 1970	20	0	20	0	-	-	-	-	0
November 9, 1970	20	10	0	60	-	-	-	-	0
February 9, 1971	-	-	-	-	-	-	-	<2	-
June 22, 1971	10	25	10	<10	-	<0.02 <sup>a</sup>	<10	<2	10
November 15, 1971	33 <sup>a</sup>	140 <sup>a</sup>	<10 <sup>a</sup>	<10 <sup>a</sup>	-	<0.02 <sup>a</sup>	<10 <sup>a</sup>	-	<1

Sample Site Description - Raisin River at the dock 500 feet upstream from the Environmental Research Center Building in Monroe.

## Key:

\* = questionable results

a = value represents total concentration of heavy metal (dissolved and particulate)

0 = chemical parameter is less than analytically detectable level

- = chemical parameter not measured

1 = WRC Interstate Rivers Monitoring Program Metal Analysis, 1963-1971, Michigan Water Resources Commission, Stevens T. Mason Bldg., Lansing, Michigan.

Table A14. The dissolved heavy metal concentrations in micrograms per liter (ppb) in water samples taken from the Rouge River at Zug Island, Michigan between 1969 and 1971. (1)

Date	Copper	Zinc	Nickel	Chromium Hexavalent	Chromium Trivalent	Mercury	Lead	Arsenic	Antimony	Cadmium
January 24, 1969	10*	40*	10*	0*	-	-	-	-	-	-
February 13, 1969	10*	30*	10*	100*	-	-	-	-	-	-
March 10, 1969	10*	20*	20*	50*	-	-	-	-	-	-
April 9, 1969	0	50	0	0	-	-	0	-	-	0
May 5, 1969	0	70	-	-	-	-	-	-	-	-
June 18, 1969	-	20	-	0	10	-	-	-	-	-
July 7, 1969	0	0	0	0	-	-	-	-	-	0
September 15, 1969	0	20	0	-	-	-	-	-	-	0
October 10, 1969	0	180	0	0	-	-	0	-	-	0
November 17, 1969	0	50	0	0	-	-	0	-	-	0
January 21, 1970	0	30	0	0	-	-	0	-	-	0
February 16, 1970	10	20	0	0	-	-	-	-	-	0
March 16, 1970	50	30	0	0	-	-	-	-	-	0
April 13, 1970	0	17	0	0	-	-	-	-	-	0
May 5, 1970	0	8	0	0	-	-	-	-	-	0



Table A14 (con't.)

June 4, 1970	0	29	0	0	-	-	-	-	0
October 12, 1970	10	0	0	0	-	-	-	-	0
November 9, 1970	0	0	0	0	-	-	-	-	0
February 9, 1971	-	-	-	-	-	-	-	<2	-
June 22, 1971	<10	15	<10	<10	-	<0.02 <sup>a</sup>	<10	<2	<1
October 14, 1971	12 <sup>a</sup>	150 <sup>a</sup>	<10 <sup>a</sup>	-	-	<0.02 <sup>a</sup>	25 <sup>a</sup>	-	<1 <sup>a</sup>

Sampling Site Description- Rouge River on D, T&I Railroad Bridge at Short Cut Channel, southern end of Zug Island.

Key:

- \* = Questionable results.
- a = Value represents total concentration of heavy metal (dissolved and particulate)
- 0 = Chemical parameter is less than the analytically detectable level.
- = Chemical parameter not measured.

(1) WRC Interstate Rivers Monitoring Program Metal Analysis, 1963-1971, Michigan Water Resources Commission, Stevens T. Mason Bldg., Lansing, Michigan.

Table A15. The dissolved heavy metal concentrations in micrograms per liter (ppb) in water samples taken from the Maumee River and North Maumee Bay Basins between 1963 and 1971. (1)

Sampling Site Location	Date	Zinc	Nickel	Chromium (Hexavalent)	Copper
1. E. Branch of St. Joseph River @ Williams County Rd.- 9 Ohio, ~ 1/2 mile S of Mich Ohio state line	8/28/69 5/12/70 9/08/70	0 10 0	0 0 0	0 0 0	0 0 0
2. Unnamed tributary to the E. branch of the St. Joseph River at Miner Rd. downstream from Pittsford	8/28/69 5/12/70 9/08/70	0 0 10	0 0 0	0 0 0	0 0 0
3. Silver Creek @ Territorial Rd. ~ 1/2 mile N or Mich - Ohio state line	8/28/69 5/12/70 9/08/70	0 10 0	0 0 0	0 0 0	0 0 0
4. Meyers Drain at Territorial Rd. ~ 1 mile SE of Waldron and in 300 ft. N of the Mich - Ohio line	5/12/70 9/08/70	15 10	0 0	0 0	0 0
5. Garrison Drain at State St. in Hudson	4/24/69 8/28/69	100 500	0 0	20 3300	0 0
6. Bean Creek at Main Street at Morenci ~ 3/5 mile N of Mich-Ohio line	5/12/70 9/08/70	30 120	0 0	0 20	0 0
7. Bean Creek @ Fulton County Rd.-T Ohio ~ 9/10 mile S of Mich-Ohio line	5/12/70 9/08/70	100 10	0 0	10 0	0 0
8. Silver Creek @ M-156 in Morenci ~ 1/10 mile W of Mich-Ohio line	9/08/70	16	0	0	0
9. N Ten Mile Creek @ Hicker Rd. 1/2 mile N of Mich- Ohio line	4/23/69 8/26/69 5/12/70	20 30 10	0 0 0	0 0 0	0 0 0
10. Halfway Creek @ Smith Rd. 1/2 mile N of Ohio- Mich line	4/23/69 8/26/69 5/12/70 9/08/70	20 10 0 10	0 0 0 0	0 0 0 0	0 0 0 0

Table A15 (con't.)

11. Halfway Creek @ State Line Rd. In Lucas Co., Ohio, 1/10 mile S of Mich-Ohio line	4/23/69	20	0	0	0
	8/26/69	20	0	0	0
	5/12/70	10	0	0	0
	9/08/70	10	0	0	0
12. Halfway Creek at Alternate US-24 2/5 mile N of Mich- Ohio line	4/23/69	40	50	0	0
	8/26/69	20	0	0	0
	5/12/70	0	0	0	0
	9/08/70	0	0	0	0
13. Silver Creek adjacent to Penn-Central RR tracks 75 yards upstream from its confluence with Halfway Creek 1/5 mile N of Mich- Ohio line	6/17/68	0	0	0	0
	4/23/69	20	0	0	0
	9/08/70	10	0	0	0
14. Ottawa River at Alt. US-24 Bridge in Lucas Co., Ohio 3/5 mile S of Mich-Ohio state line	4/23/69	20	0	0	0
	8/26/69	30	0	0	0
	5/12/70	0	0	0	0
	9/08/70	0	0	0	0
15. Shantee Creek @ Alternate US-24 1/5 mile N of Mich- Ohio state line	4/23/69	20	0	0	0
	8/26/69	30	0	0	0
	5/12/70	10	0	0	0
	9/08/70	0	0	0	0

(1) WRC Interstate Rivers Monitoring Program Metal Analysis, 1963-1971,  
Michigan Water Resources Commission, Stevens T. Mason Bldg., Lansing,  
Michigan.

APPENDIX - B

IMPACT OF WASTEWATER ON SOILS

IMPACT OF WASTEWATER ON SOILS

by

Boyd G. Ellis  
A. Earl Erickson  
Bernard D. Knezek  
Arthur R. Wolcott

Department of Crop and Soil Science

Thomas G. Bahr  
Project Coordinator

Institute of Water Research

\*Technical Report No. 30

Institute of Water Research  
Michigan State University  
East Lansing, Michigan 48823

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## SECTION I - INTRODUCTION

### Purpose

This report was prepared in accordance with contractual obligations between the U.S. Army Corps of Engineers, Detroit District and the Institute of Water Research, Michigan State University, Contract Number, DACW-35-72-C-0031. The purpose of the overall study is to describe the environmental impacts of a variety of wastewater management alternatives proposed for the Southeast Michigan area. One such alternative is the irrigation of soils with partially treated wastewater. Details of this particular alternative are given in "Interim Technical Report #1, Southeastern Michigan Wastewater Study," U.S. Army Corps of Engineers, Detroit District, 31 July, 1972 prepared by Dow Engineering, Inc., Midland, Michigan. The Dow Study concentrated on designing a system to allow soil and vegetation to remove pollutants from wastewater prior to their collection in an under-drain system and ultimate discharge into receiving waters. The details of the design were to be sufficient enough to allow accurate cost estimates and estimates of system performance.

Using the Dow Engineering, Inc. design we were able to undertake a detailed analysis of the environmental impacts of the soil disposal alternative. The analysis presented herein is intended to provide the Corps of Engineers with information necessary to more fully assess the relative advantages and disadvantages of terrestrial disposal systems. Although this report is specifically aimed at waste disposal problems of Southeastern Michigan, its scope is broad enough and many discussions are general enough to be of value to other regions sharing similar soil and climatic conditions.

## Approach

Before the impact of wastewater on a soil system can be accurately assessed it is essential to know the chemical profile of the applied wastewater and the general chemical, physical and biological features of the soil system prior to wastewater application. It is significant to point out that the wastewater applied to the soil system will be partially treated, having first been passed through a lagoon system. The lagoons proposed for this alternative are extremely large and there exists very little information on the nature of the effluent that would arise from a system of this magnitude. The best information we could find on the anticipated quality of this partially treated water was provided by the Corps of Engineers in their 1971 feasibility study. This information is shown in Table 1 and formed the basis of our assessments for this report. Unfortunately values for the expected concentrations of calcium, magnesium, sodium and potassium were not included in this earlier report. These constituents are important in chemical soil dynamics and concentrations of these cations were required to complete some of the calculations that had to be made for this report. They were, thus, estimated on the basis of expected incremental increases in concentration that one would normally find in following these constituents through a single use of water in a large city such as Detroit. It was further assumed that these cations would be conserved, the lagoon systems being largely ineffective in their removal. These estimates were done by the Institute of Water Research and are given in Table 2. The accuracy of the evaluations presented in this report will depend, in part, on the accuracy of the data in these first two tables.

The areas to be considered for the land disposal alternative are outlined in the Dow Engineering Report and include sites in Huron, Tuscola, Sanilac, Lapeer, St. Clair, Macomb, Washtenaw, Lenawee and Monroe counties. Since soil type is the critical

Table 1. Anticipated composition of wastewater for spray irrigation in the Southeast Michigan area.<sup>1</sup>

	Raw Waste Influent	Polishing Lagoon Effluent % Removal	mg/l
Suspended Solids	226.	85	34.0
Settleable Solids	129.	95	6.0
BOD	132.	90	13.2
COD <sup>2</sup>	350	80	70
Organic Nitrogen	13.3	67	4.4
NH <sub>3</sub>	7.5	67	2.5
NO <sub>2</sub>	.002	--	
NO <sub>3</sub>	.05	--	0.9
Total Nitrogen	21.5	67	7.1
Total Phosphorus	11.7	40	7.0
Oil & Grease	71	60	28.
Chlorides	184	nil	184
Fe	8.03	30	5.6
Cu	0.36	50	0.18
Ni	0.52	70	0.16
Zn	0.44	nil	0.44

<sup>1</sup>Data taken from Table C-14, Appendix C, Alternatives for managing wastewater for Southeastern Michigan. A feasibility study by the Detroit District, U. S. Army Corps of Engineers, July, 1971.

<sup>2</sup>COD value taken from Table IV-2 of the Summary Report of the above study.

Table 2. Calculated concentration of sodium, potassium, calcium and magnesium in wastewater effluent from the Detroit area.

Cation	Concentration	
	ppm	meq/l
Na	120	5.22
K	15	0.38
Ca	70	3.5
Mg	35	2.92

factor in developing design criteria for land disposal systems we concentrated on soil type rather than addressing our assessments to specific areas. The major soil types to be found in the proposed areas are listed in Table 3. No analyses were made specifically on soils in the projected areas but a considerable body of data exists on these and similar soil types. These data have been included as a point of departure and should be sufficiently accurate for the Survey Scope phase of this study. It should be stressed that soils are variable and that final design should allow for this variability. It is recommended that before actual implementation of specific land disposal alternatives, field sampling from the proposed sites be conducted.

Other design criteria such as the depth of the tile lines and the rate of water application together with the pattern of spreading should be designed to fit the individual soils. The following criteria were specified for the purposes of this report and will be used: Depth to tile = 6 feet; rate of water application = 2 inches per week; pattern of spreading will be by a central pivot system making one revolution in 5 days with a two day rest period; and the irrigation will be for 35 weeks of the year. Other

Table 3. Soil types and chemical properties of the major soil types in proposed spray irrigation areas.

Soil Series	Horizon	Texture	Depth inches	CEC	Ca	Mg	Na	K	P	Zn	C	N
				milliequivalents/100 grams								
					percent							
Conover	Ap	loam	0-7	12.1	7.5	1.2	.1	.13	20	5	1.71	.16
	A <sub>2</sub>	silt loam	7-11	11.1	6.9	1.1	.1	.15	2	2	.36	
	B <sub>1</sub>	silty clay loam	11-18	16.1	10.0	1.6	.1	.14	2	4	.31	
	B <sub>2</sub>	sandy clay	18-30	14.7	9.1	1.5	.1	.16	3	4	.18	
	C	silty clay	>30	9.9	6.1	.99	.1	.05	3	0.6	2.72	
Parkhill	Ap	loam	0-12	19	16.9	1.8	.15	.32	50	8.6	3.5	.20
	B	clay loam	12-36	13	10.4	2.4	.10	.18	45	3.9	1.2	.06
	C	loam	>36	11	8.8	2.0	.08	.18	75	2.6	1.4	
Locke	Ap	sandy loam	0-7	5.4	1.7	.16	.08	.13	5.2	2.3	1.09	.1
	A <sub>2</sub>	sandy loam	7-12	4.7	1.5	.14	.08	.13	6.3	2.3	.74	.06
	B	loam	12-22	6.1	2.0	.18	.10	.13	3	1.6	.35	
	B	loam	22-25	5.5	1.8	.16	.09	.13	7	2.5	.11	



Table 3 (con't.)

	B	loam	26-33	5.5	1.8	.16	.09	.13	23	1.7	.10
	C	sandy loam	>33	3.7	1.2	.11	.06	.09	22	.8	.10
Blount	Ap	loam	0-8	14.7	7.2	2.1	.1	.2			
	B	clay loam	8-28	11.5	5.4	1.9	.1	.1			
	C	clay loam	>28	9.9	6.1	.99	.1	.05	3	.6	
Metamora	Ap	sandy loam	0-9	5.7	3.4	.18	.05	.12	6.5	.98	.09
	A <sub>2</sub>	sandy loam	9-15	2.8	1.7	.08	.03	.12	4.5		
	B <sub>1</sub>	loam	15-24	2.0	1.2	.06	.02	.10	13		
	B <sub>2</sub>	loam	24-50	4.9	2.9	.15	.05	.09	13	.19	
	C	sandy loam	>50	2.6	1.6	.08	.03	.10	50	.15	
Capac	Ap	sandy loam	0-12	5.7	3.4	.17	.05	.12	6.5	1	.1
	B	clay loam	12-34	15.4	9.5	1.5	.1	.14	2.5		
	C	loam	>34	9.9	6.1	.99	.1	.05	3		
Brookston	Ap	loam	0-8	25	17.5	1.65	.15	.11	21	3.8	3.5 .34
	B <sub>1</sub>	silty clay	8-12	21	13.9	2.50	.18	.10	36	3.6	1.8 .19

Table 3 (con't.)

Fabi	B <sub>2</sub>	silty clay	12-18	15	9.9	1.78	.13	.11	54	.9	.76	.08
	B <sub>3</sub>	silty clay	18-36	14	9.2	1.67	.12	.10	69	.9	.41	
	C	loam	>36	13	8.6	1.55	.11	.11	51	.5	.27	
Fabi	Ap	sandy loam	0-8	4.9	3.2	.06	.06	.16	11		.84	.07
	B <sub>1</sub>	sandy loam	8-12	7.7	5.1	.1	.09	.17	7		.31	
	B <sub>2</sub>	sandy clay loam	12-21	9.9	6.5	.13	.12	.17	10			
	B <sub>3</sub>	sandy loam	21-34	4.7	3.1	.06	.06	.14	14			
	C	sand	>34	1.5	1.0	.02	.02	.01	8			

alternatives will be considered in the report to allow for evaluation of lesser spreading rates.

### Background

Numerous recent reviews and bibliographies have been directed toward collating a vast literature related to natural and agricultural systems into a waste management frame of reference (Ellis, 1972; McGauhey and Krone, 1967; CRREL, 1972; WRSIC, 1972; Driver, *et al.*, 1972). Cycles of carbon, nitrogen and other mineral nutrients have been described in many contexts (Eglinton, 1969; Brock, 1966; Campbell and Lees, 1967; Burges, 1967; Bollen, 1959; Deliwiche, 1965; Janssen, 1958, 1971; Alexander, 1971). In Figure 1, the organic phase of soils is depicted as a detritus system in which the major flow of carbon follows the major flow of energy captured by photosynthetic plants and disengaged by decomposition of dead plant tissues by soil heterotrophs (Odum, 1969; Pomeroy, 1970).

In most soils, the biomass represented by dead tissue and living decomposers is small relative to the mass represented by internally, cycling organic phase components. These cycling components are subdivided in Figure 1 into three major groups: exometabolites, fulvic complexes and humic complexes. This classification conforms to current views regarding sequences involved in humic acid synthesis (Stevenson and Butler, 1969; Hurst and Burges, 1967; Kononova, 1966). Susceptibility to extraction of the three groups of compounds is outlined in Figure 2, and important physical and chemical properties are compared in Figure 3. Similar classifications have been employed in characterizing organics in sewage effluents (McGauhey and Krone, 1967; Rebhun and Manka, 1971).

The flow of carbon through the plant-soil system is accompanied by cyclic immobilization and mineralization of mineral nutrients, as well as of non-nutrient mineral species. However, net flows of minerals do not necessarily parallel net flows of

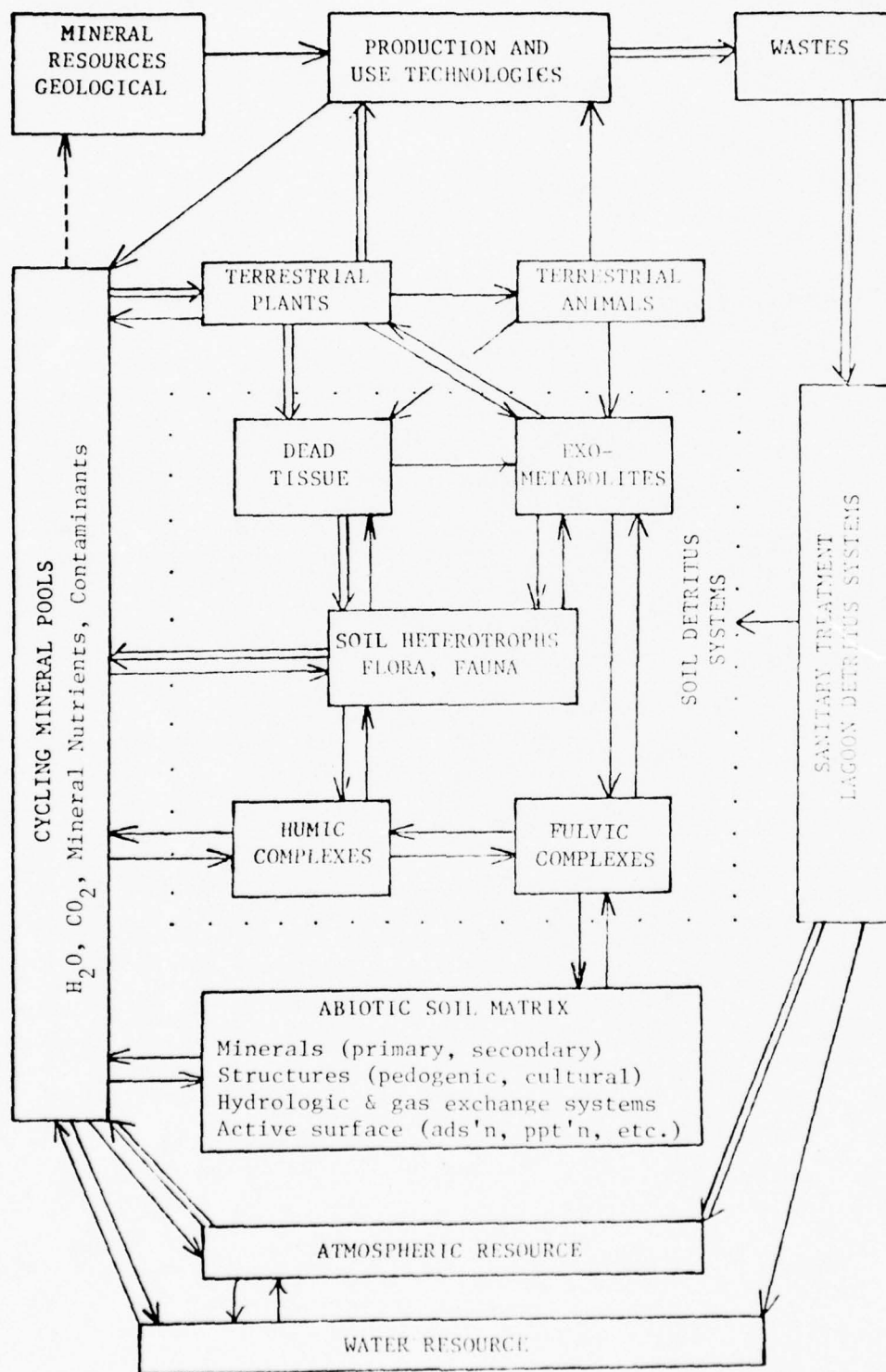


Figure 1. Carbon Flows and Interfaces with Soil Detritus Systems.

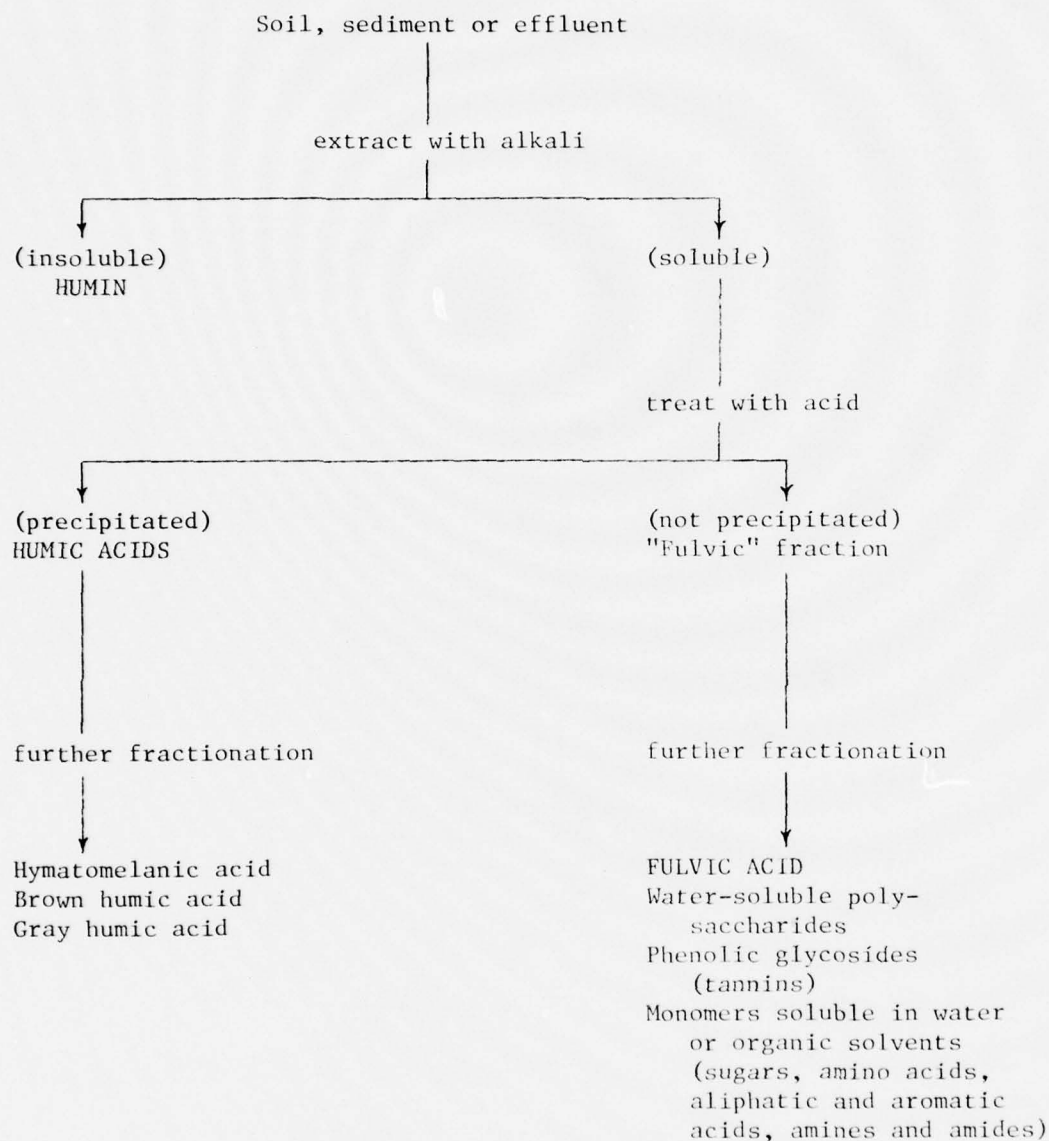
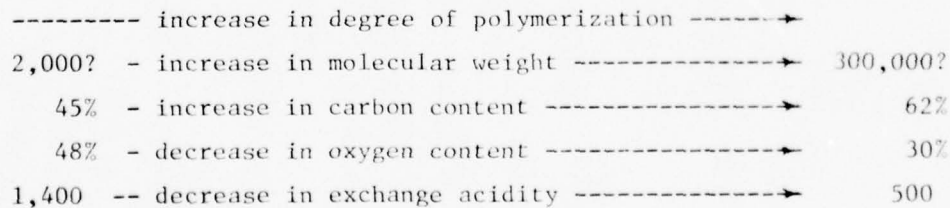


Figure 2. Fractionation of organic phase components of soils, sediments, waste effluents. Adapted from Stevenson, 1965a, Stevenson and Butler, 1969; Rebhun and Manka, 1971; Arthur D. Little, Inc., 1971.





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carbon, except at homeostatic equilibrium. Each must be considered individually.

The compartmental classification shown in Figure 1 does, nevertheless, provide a framework for relating physical and chemical phenomena to biological activities and their derivative organic phase systems. These activities and systems differentiate soils from geological mineral deposits.

## PART 1 -- CHEMICAL REACTIONS IN THE SOIL

### Sodium, Calcium, Potassium and Magnesium

These elements are all found in natural soil systems and are seldom limiting for growth in aquatic systems although total salt concentrations can, at times, reach sufficiently high levels in terrestrial systems to limit growth of certain plants. But the levels of salts added in sewage effluents are not sufficiently high to affect more than the most sensitive plants and should not present any difficulties to agricultural crops grown in Michigan. The most sensitive crop presently grown in the proposed irrigation area is field beans and these are not likely to be one of the crops grown for other reasons. Corn, reed canary grass and other proposed crops should encounter no salinity hazard.

On the other hand, soils themselves may be susceptible to degradation due to excess adsorption of monovalent ions in the wastewater. There are many equations which have been used to predict the change in adsorption ratio which will occur. One of these involves the use of the sodium-adsorption-ratios (SAR) and it has been applied to the data in Table 2. SAR is defined by the following equation:

$$SAR = \frac{(Na)_{meq/l}}{\sqrt{\frac{(Ca + Mg)_{meq/l}}{2}}}$$

Table 4 contains calculated exchangeable sodium percentages and exchangeable sodium plus potassium percentages. Although quite insignificant in the wastewater we are considering here, potassium can sometimes be quite important. It was included as a separate calculation to show its relative insignificance. The increasing levels of sodium were included to give a quick reference in the event that a practice comes under consideration which would increase the level of sodium in the wastewater (for example,

Table 4. The effect of increasing sodium concentration in waste-water on the SAR, SKAR, Exchangeable Sodium (ESP) and Sodium + Potassium Percentage [E(S+K)P<sup>1</sup>].

Na Conc	SAR	SKAR	ESP <sup>1</sup>	E(S+K)P <sup>1</sup>
ppm			----- % -----	
120	2.95	3.17	6	6.6%
150	3.69	3.90	7	7.5
200	4.92	5.13	9	9.3
250	6.15	6.37	10.5	10.8
300	7.38	7.60	12	12.2
350	8.61	8.83	14	14.2
400	9.84	10.06	15.5	15.7
450	11.07	11.29	16.5	16.7
500	12.30	12.52	18.0	18.2

<sup>1</sup>Estimated from figure 23 of USDA Handbook 60.

industrial discharge or the practice of applying sodium chloride to streets in large quantities in the winter months and not separating the storm sewer discharge from the septic discharge). The direct effect of increasing sodium percentages is on the soil structure and will be discussed later in this report.

The exchangeable sodium percentage reached is independent of the soil type if the wastewater is applied long enough to reach steady state. Soils containing free calcium carbonate, however, will require much more time to reach this steady state. It is expected that soils without free calcium carbonate will reach equilibrium in 2 to 3 years if two inches per week are applied but at lower application rates this length of time will increase. The influence of increasing levels of exchangeable calcium may become evident during the first year of application before a steady state situation is reached.

Prior to reaching a steady state the drainage water from the irrigation area should be somewhat lower in sodium than the incoming wastewater and correspondingly higher in calcium and magnesium. Once steady state is established the drainage water will be similar to the applied wastewater with respect to these elements.

### Phosphorus

A description of how phosphorus cycles through the soil system is presented in Figure 4. Mineral resources are used in preparation of fertilizer from which they are fed into cycling mineral pools and incorporated into plants, and large quantities of phosphorus also are used in industry and by the general populace (detergents are a good example). Waste flows at the present time carry phosphorus from both of these sources. Spray irrigation of these wastes would mean that large amounts of phosphorus would be carried into the soil and adsorbed on the soil matrix and discharge into the water resource would occur only after passing through this complex. Although phosphorus will enter into soil organic reactions, these would be of lesser importance.



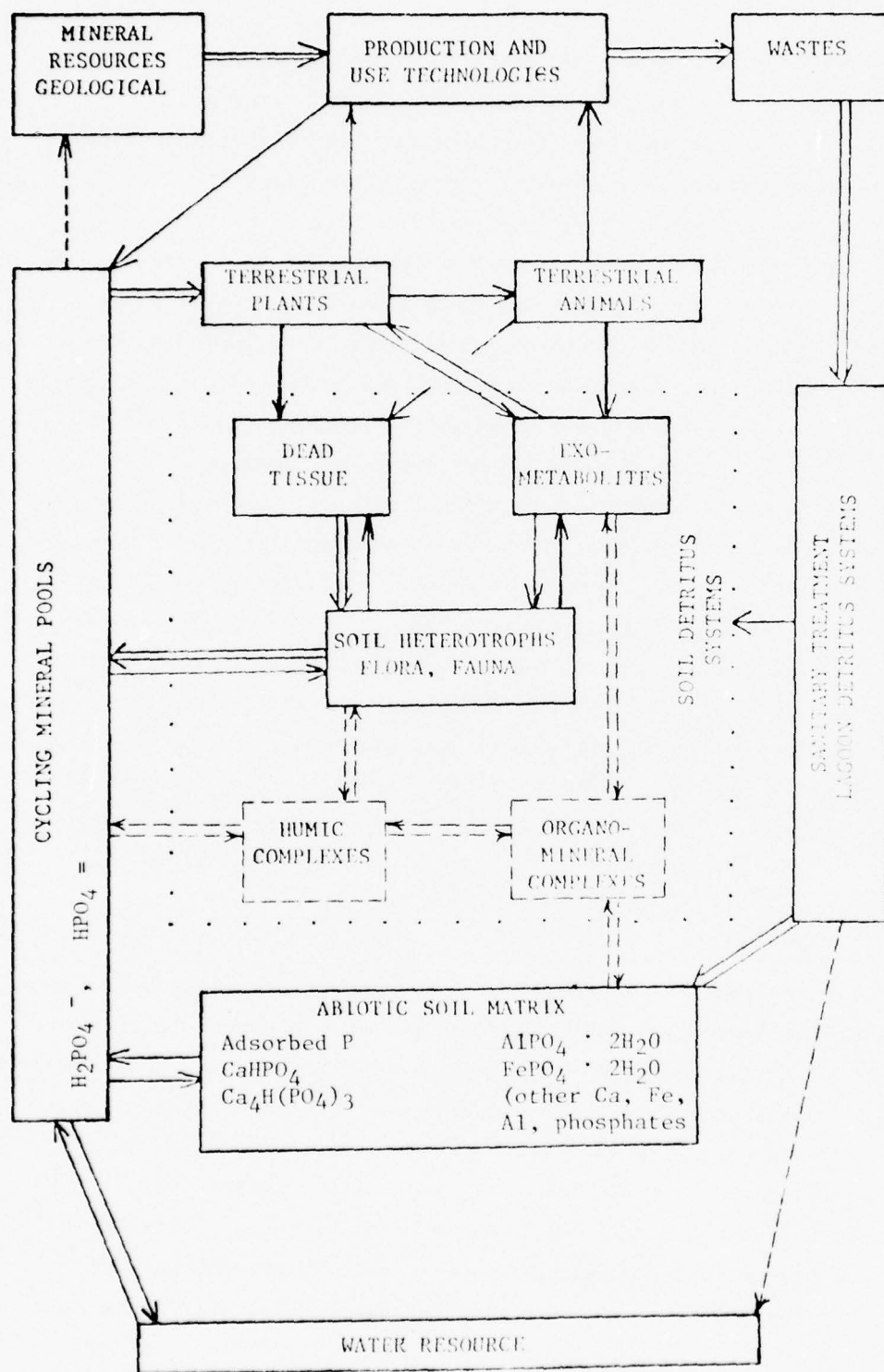


Figure 4. Phosphate flow and interfaces with soil detritus systems in land disposal of municipal sanitation effluents.

Removal of phosphorus from wastewater is one of the most critical factors in the design of any waste treatment system. The soil system is able to accomplish this through two mechanisms. First, by the incorporation of phosphorus into living organisms, particularly plants. The living filter concept, as reported by research workers at Pennsylvania State University (Parizek, et al., 1967), is a good example of such a system. In 1963 their crops (red clover, alfalfa, corn and wheat) removed an average of 23 pounds of phosphorus per acre. Using this value as an average quantity of phosphorus removed by crops it is calculated that the system could utilize 15.77 inches of water per acre per year and produce a balanced system with respect to phosphorus. The concentration of phosphorus in the drainage water should be about 0.05 mg/l for the soils under consideration. But at higher application rates the soil would be expected to adsorb or precipitate some of the phosphorus in solution.

The second method of removal of phosphorus is through adsorption reactions. Precipitation should not be expected to occur in land disposal systems. The exact quantity of phosphorus that can be adsorbed by soils is a matter of some debate. Extremely high values have been reported by Toth and Bear (1947) from a laboratory study involving acid treatment of soils. The use of such procedures or estimates which are based on the total iron and/or the aluminum content of soils is very misleading. But such estimates have been made.<sup>1</sup> The use of adsorption isotherms to measure the quantity of phosphorus adsorbed from soil suspensions in which the other ion content and pH are similar to those which will be encountered in the field should give more reasonable estimates.

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<sup>1</sup>Bauer, W. J. 1972. Presented at symposium on recycling treated municipal wastewater and sludge in paper titled "Large wastewater irrigation systems: Muskegon County, Michigan and Chicago metropolitan region.

The Langmuir adsorption isotherm is the most useful of the adsorption isotherms because of the prediction of an adsorption maximum and the ease with which the determinations can be made. The use of this isotherm for predicting phosphorus adsorption by soils has been reviewed earlier (Ellis, 1972) and as is pointed out in this paper it is likely a conservative estimate. Laboratory data have shown that soils will regenerate their capacity to fix phosphorus when once saturated if allowed to "rest" for a period of three months although it is not known if they can go through more than one cycle of regeneration.

There is little data to test the validity of the Langmuir equation to the field. The experiment at Pennsylvania State University has been running for 10 years, but even in areas of high application in this study, the quantity of phosphorus applied, beyond that removed by the crop, would not be expected to saturate the top one foot.

Data for the Langmuir adsorption isotherm for soil types similar to those proposed for the Southeast Michigan area have been measured and are reported in Table 5. It should be particularly noted that the adsorption maxima calculated from Langmuir adsorption isotherms are for high concentrations in solution. The percentage of this maximum that can be used for adsorbing from solutions at 7 mg/liter have been calculated and all data are corrected to this value. This of course means that the level of phosphorus in the effluent when this maximum is reached would be 7 mg/liter. But by not utilizing the last foot of soil in the calculations, the values arrived at should give adequate protection against excessive levels of phosphorus in the drainage effluent.

To allow for rapid evaluation of the effect of phosphate removal by systems of various designs the number of years that the soil would effectively remove phosphorus was calculated and is shown in Table 6. To make these calculations it was assumed that cropping would remove 25 pounds of phosphorus per acre per year. As pointed out earlier, this should be a reasonable

Table 5. Phosphorus adsorbing capacities of the major soil types in the proposed area for spray irrigation.

Soil Series	Horizon	Texture	Depth	K <sup>1</sup>	b <sup>2</sup>	Available Fraction <sup>3</sup>	P adsorbing capacity		
							Acre Foot	Acre Horizon	Acre-3 Feet
							----- lbs P -----		
				Kx10 <sup>-4</sup>	mg/100 g	%			
Conover	A <sub>p</sub>	loam	0-7	2.79	10.75	86	334	306	
	A <sub>2</sub>	silt loam	7-11						
	B <sub>1</sub>	silty clay loam	11-18	8.06	20.25	95	784	1241	1800
	B <sub>2</sub>	sandy clay	18-30						
	C	silty clay	30	5.25	13.5	92	506	253	
Parkhill	A <sub>p</sub>	loam	0-12	6.21	5.0	93	168	168	
	B	clay loam	12-36	2.30	8.8	84	301	602	770
	C	loam	36	6.49	16.7	94	640		
Locke	A <sub>p</sub>	sandy loam	0-7	3.22	12.7	88	403	403	
	A <sub>2</sub>	sandy loam	7-12						

Table 5 (con't.)

	B	loam	12-33	5.02	11.1	92	416	728	1231
	C	sandy loam	33	3.74	11.0	89	399	100	
Metamora	A <sub>p</sub>		0-9	2.66	10.1	86	314	392	
	A <sub>2</sub>		9-15						
	B <sub>1</sub>		15-24	1.01	25.0	69	703	1230	1622
	B <sub>2</sub>		24-50						
	C		50	3.54	7.6	89	276		
Capac	A <sub>p</sub>	sandy loam	0-12	2.66	10.1	86	314	314	
	B	clay loam	12-34	8.06	20.25	95	784	1437	1835
	C	loam	34	5.25	13.5	92	506	84	
Brookston	A <sub>p</sub>	loam	0-8	5.71	18.6	93	624	624	
	B <sub>1</sub>	silty clay	8-12						
	B <sub>2</sub>	silty clay	12-18	6.56	16.0	94	613	1226	1850



Table 5 (con't.)

B <sub>3</sub>	silty clay	18-36			
C	loam	36	2.93	14.1	86
					500

1, <sup>2</sup>Constants from Langmuir equation. K is related to bonding energy; b is the P adsorbing maximum.

<sup>3</sup>Fraction of the adsorbing maximum that will be used at 7 ppm P.

Table 6. The length of time that soils will adsorb phosphate based on loading rate and depth to tile drain.

Conover										
Depth to Drain	35 weeks/year					52 weeks/year				
	0.5	1	2	4	in/week	0.5	1	2	4	in/week
feet	years									
3	900	62	22	9.4		112	32	13	5.9	
4	1153	80	28	12		144	40	17	7.6	
5	1406	97	34	15		176	49	20	9.3	
Parkhill										
Depth to Drain	35 weeks/year					52 weeks/year				
	0.5	1	2	4	in/week	0.5	1	2	4	in/week
feet	years									
3	386	27	9.3	4		48	13	5.5	2.5	
4	705	49	17	7.4		88	25	10	4.7	
5	1025	71	25	10.7		128	36	15	6.8	
Locke										
Depth to Drain	35 weeks/year					52 weeks/year				
	0.5	1	2	4	in/week	0.5	1	2	4	in/week
feet	years									
3	615	42	15	6.4		80	22	9	4	
4	815	56	20	8.5		102	29	12	5.4	
5	1014	70	24	10.6		127	36	15	6.7	

Table 6 (con't.)

Metamora										
Depth to Drain	35 weeks/year					52 weeks/year				
	0.5	1	2	4	in/week	0.5	1	2	4	in/week
feet										
	years									
3	811	56	20	8.5		101	28	12	5.4	
4	949	65	23	9.9		119	33	14	6.3	
5	1087	75	26	11.4		136	38	16	7.2	

Capac and Brookston										
Depth to Drain	35 weeks/year					52 weeks/year				
	0.5	1	2	4	in/week	0.5	1	2	4	in/week
feet										
	years									
3	917	63	22	9.6		115	32	13	6.1	
4	1170	81	28	12.2		146	41	17	7.7	
5	1423	98	34	15		178	50	20	9.4	

assumption. However, if high rates of spreading (i.e., 2 or 4 inches per week) should create aeration problems and subsequent reducing conditions, resulting in poor growth of crops, this value would not be realized. Likewise, if the crop is not removed by harvesting it would not be realized. It should be pointed out here that animals would remove only about 15 percent of the 25 pounds or 3.75 pounds phosphorus per acre per year if allowed to graze on grass. Under such a management system the calculations would need to be adjusted.

Three depths of soil were included in the calculations: 3, 4 and 5 feet. In the case of depth it is really the depth to saturated conditions that represents the critical factor. The design criteria gave 6 feet to the depth of the tile lines. Leaving one foot of soil which should never become saturated with phosphorus (to assure low concentrations of phosphorus in the drainage water) the maximum depth that can be utilized is five feet. However, between tile lines the depth to saturated conditions will be less than six feet, thus, the other two depths were included for comparisons. Combining two periods of spreading (35 weeks/year and 52 weeks/year) with four rates (0.5, 1, 2, and 4 inches per week) gives eight different loading rates which have been calculated and are included in the table. Other rates of loading can be easily calculated if needed.

Data are not presently available for all soil types in the proposed spray irrigation site. However, the Blount, and Fabius soils should be intermediate in their ability to adsorb phosphorus. They are also much smaller in acreage in the proposed area than other soils types. To interpret Table 6, it is assumed that the system should be designed to last for a minimum of 25 years.

The soil with the lowest phosphorus adsorbing capacity is the Parkhill loam. No more than one inch per week for 35 weeks of the year should be considered on this soil. It should also be pointed out that the calcareous nature of the subsoil for this soil type will likely lead to somewhat higher phosphorus concentration in the drainage effluent. But they should be well below

1 ppm phosphorus for the first 25 years if not applied in quantities greater than 1 inch per week for 35 weeks of the year. If applied at 2 inches per week the system should pass phosphorus in excess of 5 ppm between 10 and 17 years after starting depending on the average depth to saturated soil. Four inches per week would have such a short life that it cannot be seriously considered for the Parkhill or any other of the soil types involved.

The Locke soil is the next lowest in adsorbing capacity. It could be expected to handle about 1.5 inches of wastewater per week for 35 weeks of the year.

Conover, Metamora, Capac and Brookston are all similar. They may utilize nearly 2 inches of wastewater per week for 35 weeks of the year. After about 25 years of receiving wastewater at this rate their adsorbing capacity would be saturated.

#### Chloride

The chloride ion is not adsorbed by soils to any extent. The actual quantity of chloride being applied appears to be excessive (see Table 7), but since it leaches it is not critical.

Table 7. Quantity of chloride that will be applied to each acre spray irrigated.

Rate of Application	Weeks of Application	
	35	52
inches/week	Pounds Cl applied/acre/year	
0.5	729	1083
1.0	1459	2167
2.0	2918	4334
4.0	5836	8669



Since the soil does not adsorb chloride the total quantity of chloride applied is of lesser importance providing sufficient water is applied to leach the soil (one inch per week or greater). The concentrations of chloride in the drainage effluent is expected to be near that being spray irrigated. Most of the crops under consideration will tolerate this level of chloride including hay and corn.

#### Sulfate

No values for sulfate were available, thus, it is not possible to calculate the quantities that would accumulate in soil. Sulfate adsorption by soils is discussed by Ellis (1972b). It should be noted that sulfate can be reduced in soils under anaerobic conditions and this could occur at an application rate of 2 or 4 inches per week.

## PART 2 -- HEAVY METAL REACTIONS IN THE SOIL

Heavy metals cycle in many forms from geochemical sources through plants, animals and man and in geological time eventually return to the sources. Allaway (1968) and Bowen (1966) have discussed trace element cycling in depth. Reactions of metals in soils have been reviewed by Ellis and Knezek (1972) and Hodgson (1963). The monograph by Bear (1964) on soil chemistry is a good reference text for general information on chemical reactions in soils. For projections of expected chemical levels and effects in plants and soils the publication edited by Chapman (1966) titled "Diagnostic Criteria for Plants and Soils" is an exceptionally useful text due to separate discussions about plant and soil factors on each nutrient.

For the purposes of this discussion the term heavy metal will include arsenic, boron, and selenium which normally are not chemically classified as metals. The data given in Table 8 show the pounds of metal that would be added to each acre of land at 0.5, 1, 2 and 4 acre inches of effluent application per week. The data are calculated on the basis of 35 and 52 weeks of spraying time each year respectively. The amount of metal added under a particular spray regime (Table 8) can be compared with the average total amount of the metal that is found in an acre furrow slice of soil (Table 9) (acre furrow slice = one acre of mineral soil to a depth of six inches weighing approximately 2,000,000 pounds per acre). Although most metals may be considered fairly evenly distributed through the soil profile, there can be important changes in composition with depth (Figure 5). As a result, the average values discussed will pertain to the acre furrow slice unless otherwise specified. The accumulation of metals in the soil with respect to the "normal residual" content of metals already present in the soil (Table 9) gives one measure of the impact that additions may have on the soil system. Such information is of limited value, however, because the fixation, leaching and plant uptake potentials are governed by the form in which

Table 8. The quantity of metals that would be added to each acre of soil under alternative rates and duration of spray application.

Effluent Composition	35 wk. spray/year				52 wk. spray/year				
	Acre in. water/wk.				Acre in. water/wk.				
	0.5	1	2	4	0.5	1	2	4	
mg metal/l	Pounds Metal Added/Acre/Year								
Arsenic <sup>c</sup>	.05	0.20	0.40	0.80	1.60	0.29	0.59	1.18	2.3
Boron <sup>b</sup>	1.0	3.95	7.90	15.8	31.6	5.85	11.7	23.4	46.8
Cadmium <sup>b</sup>	.015	0.06	0.12	0.24	0.48	0.09	0.18	0.36	0.72
Chromium <sup>b</sup>	.14	0.55	1.11	2.22	4.44	0.82	1.64	3.28	6.56
Cobalt <sup>c</sup>	.20	0.79	1.58	3.16	6.32	1.17	2.35	4.70	9.40
Copper <sup>a</sup>	.18	0.74	1.44	2.88	5.76	1.05	2.11	4.22	8.44
Iron <sup>a</sup>	5.6	22.0	44.1	88.2	176.4	32.7	65.5	131.0	262.0
Lead <sup>b</sup>	.03	0.14	0.24	0.48	0.96	0.17	0.35	0.70	1.40
Manganese <sup>b</sup>	.20	0.79	1.58	3.16	6.32	1.17	2.35	4.70	9.40
Mercury <sup>b</sup>	.01	0.04	0.08	0.16	0.32	0.06	0.13	0.26	0.52

Table 8 (con't.)

Nickel <sup>a</sup>	.16	0.63	1.26	2.52	5.04	0.94	1.88	3.76	7.52
Selenium <sup>c</sup>	.05	0.20	0.40	0.80	1.60	0.29	0.59	1.18	2.36
Zinc <sup>a</sup>	.44	1.74	3.48	6.96	13.92	2.58	5.16	10.32	20.64

<sup>a</sup>Data taken from Table C-14, Appendix C, Alternatives for managing wastewater for Southeastern Michigan. A feasibility study by the Detroit District, U. S. Army Corps of Engineers, July, 1971.

<sup>b</sup>Christman, Russel F. (Project Coordinator) Assessment of the effectiveness and effects of land disposal methodologies of wastewater management. Final report in fulfillment of contract numbers DACW 73-73-C-00 through 0043 for the Office of the Chief of Engineers, U. S. Army Corps of Engineers, January 14, 1972.

<sup>c</sup>Reed, S. C. (Coordinator) Wastewater management by disposal of the land. Special Report 171, Cold Regions Research and Engineering Laboratory, U. S. Army Corps of Engineers, Hanover, N. H., May, 1972.

Table 9. Quantity of total metal increase compared to an average soil composition during the projected life of the effluent spray irrigation system.<sup>a</sup>

	Soil Metal Composition Average Range lbs./acre furrow slice	Years of Effluent Spraying				Soil Metal Increase Year 1 %
		1	5	15	25	
		lbs. metal added/acre				
Arsenic	12 (0.2-80)	0.80	4.0	12.0	20.0	6.7
Boron	20 (4-200)	15.8	79.0	237	395	79
Cadmium	0.1 (0.02-14)	0.24	1.2	3.6	6.0	240
Chromium	200 (10-6000)	2.22	11.1	33.3	55.5	1.1
Cobalt	16 (2-80)	3.16	15.8	47.4	79.0	19.7
Copper	40 (4-200)	2.88	14.4	43.2	72.0	7.2
Iron	38000	88.2	441	1323	2205	0.23
Lead	20 (4-400)	0.48	2.4	7.2	12.0	2.40
Manganese	1700 (200-8000)	3.16	15.8	47.4	79	0.19
Mercury	0.8 (0.03-0.8)	0.16	0.8	2.4	4.0	20.0
Nickel	80 (20-2000)	2.52	12.6	37.8	63	3.1
Selenium	1.0 (0.2-4.0)	0.80	4.0	12.0	20	80.0



Table 9 (con't.)

Zinc	100	(20-600)	6.96	34.8	104.4	174	7.0
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<sup>a</sup>Based on the stated effluent composition applied at two acre inches per week for 35 weeks each year.  
Assumes metals will be retained in the top six inches of soil

<sup>b</sup>Allaway (1968), Bowen (1966)

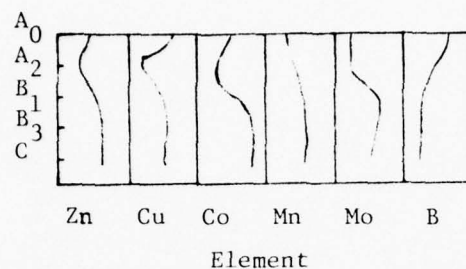


Figure 5. Distribution of micronutrients in soil profiles of near modal podzols

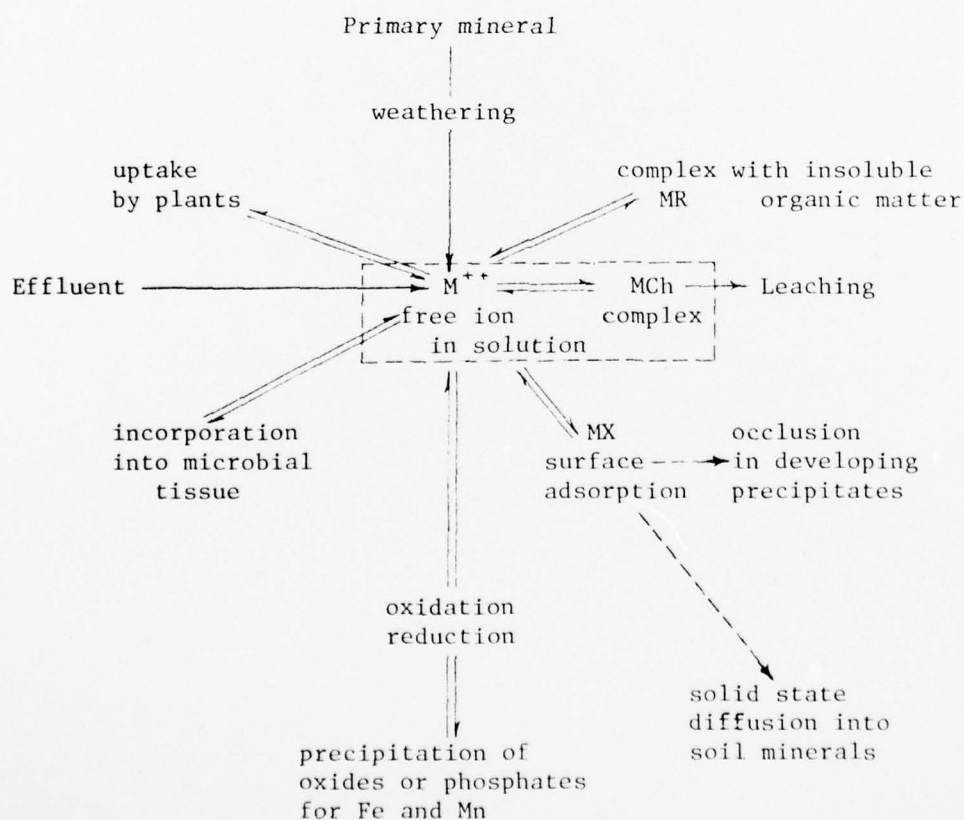


Figure 6. Fate of metals added to soils (Hodgson, 1963).

metals exist in the soil solution. When a metal is applied to the soil, several possible reactions may occur (Figure 6).

The limits of the system must be established initially. Microbial populations may increase and change in numbers but the metal content of microbial tissue is a small part of that in the soil.

An ineffective cycling mechanism for metals is through plant uptake and crop removal. The data in Table 10 clearly show that a crop of either 104 bushels of corn grain per acre or 3 tons of corn silage (dry weight) per acre would remove one percent or less of most metals added on an annual basis. Generally the rooting zone of a plant already contains one hundred to one hundred thousand times the amount of a trace element likely to be removed by any one crop. A real potential danger in spray irrigation of a crop with metal containing effluents is in surface adherence to leaves of crops and possible foliar absorption into the plant.

Basically, the proposed system must be viewed in terms of the capacity of the soil to fix metals so that they will not be toxic to plants or soil microbes nor leach into the drainage water. Each of the elements will be discussed in terms of adding effluent at the rate of two inches per week for 35 weeks per year. Unless otherwise stated, the metal will be assumed to remain in the acre furrow slice of the soil.

#### Arsenic

Arsenates in soils compete for the same fixation sites that are utilized by phosphorus. However, the arsenates are bound in the soil with less strength than phosphates and there is a real danger of leaching arsenates through the soil profile as the phosphorus adsorbing capacity of the soil become saturated. A whole range of total arsenic values from 0.3 to over 100 pounds per acre have been reported on normal and arsenic contaminated soils. Although arsenic concentration may be increased in plant tops, there is little danger of animal toxicity. The effect of

Table 10. A projected annual balance between metal added in effluent and metal removal by cropping.

	Plant Metal Concentrations <sup>a</sup>		Annual Metal Addition to Soil <sup>b</sup>	Average Annual Metal Removal by Cropping <sup>c</sup>	
	Average Range	ppm	lbs./A	lbs./A	% Metal Added
Arsenic	1	(0.1-5.0)	0.80	0.0060	0.75
Boron	30	(5-30)	15.80	0.18	1.14
Cadmium	0.6	(0.2-0.8)	0.24	0.0036	1.50
Chromium	0.23	(0.2-1.0)	2.22	0.0014	0.06
Cobalt	0.13	(0.05-0.5)	3.16	0.0008	0.03
Copper	14	(2-15)	2.88	0.084	2.92
Iron	290		88.2	1.74	1.97
Lead	2.7	(0.1-10)	0.48	0.0162	3.37
Manganese	52	(15-100)	3.16	0.312	9.87
Mercury	.015		0.16	0.00009	0.06
Nickel	3		2.52	0.018	0.71
Selenium	0.2	(0.02-2.0)	0.80	0.0012	0.10

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ECOLOGICAL ASSESSMENTS FOR WASTEWATER MANAGEMENT IN SOUTHEASTER--ETC(U)  
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Table 10 (con't.)

Zinc	50	(8-100)	6.96	0.102	1.46
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<sup>a</sup>Allaway (1968), Bowen (1966), and other sources

<sup>b</sup>Based on two acre inches of effluent per week sprayed for 35 weeks per year

<sup>c</sup>Based on 6000 pounds of dry material per year [equivalent to 104 bushels of corn grain or about 14 tons (wet weight) of corn silage per year]

arsenic toxicity on plants is such that plant growth is limited before large amounts of arsenic are absorbed and translocated to the top (Allaway, 1968; Liebig, 1966). The concentration of arsenic in corn tops has been increased to about 3 ppm without plant toxicity (Liebig, 1966). There should be little problem with leaching, plant uptake or plant toxicity of arsenic unless the phosphorus adsorbing capacity of the soil is nearing saturation. Several recent articles (Deuel, *et al.*, 1972; Jacobs, *et al.*, 1970; Johnson and Hiltbold, 1969; Woolson, *et al.*, 1971a, 1971b) tend to confirm the soil and plant reactions of arsenic.

#### Boron

If the relatively high amounts of boron projected for effluents are applied to the soil, there may be serious leaching to the drainage water and possible plant toxicity depending upon the crop being grown. Rhoades, *et al.* (1970a) have stated that many plants are deleteriously affected by boron concentrations in the range 0.5 to 5 mg/liter. Toxic levels of soluble boron may be removed from soils by leaching, but not as fast as chloride or sulfate. Ellis and Knezek (1972) have reviewed the bonding mechanisms for boron in soils. Although the evidence is not conclusive, it must be assumed that the soil solution of the furrow slice will be essentially at equilibrium with the added water (1.0 mg/l) by the end of the first 35 weeks of application due to a high rate of boron addition in the effluent. The soil may adsorb sufficient excess boron over a period of time to actually increase the level of soluble boron above the 1.0 mg/liter limit after the first year (Rhoades, *et al.*, 1970b). Corn and most other cereal grains are semitolerant to boron while most sensitive plants, such as navy beans, will tolerate 0.7 mg/liter of boron (Richards, 1954). The soluble boron levels could approach toxic levels in the soil even for corn. A further assumption must be made that after the first year of spraying wastewater, boron will essentially be moving out in the drainage water at a concentration equal to or

greater than that which is applied depending upon evaporation and evapotranspiration rates of water loss. Bingham, et al. (1971) have shown boron to be bound to the soil aluminum and iron fraction of the soil.

#### Cadmium

The chemistry of cadmium is similar to that of zinc and the soil bonding mechanisms of zinc have been recently reviewed by Ellis and Knezek (1972). Even though the total amount of cadmium in the soil will be increased several fold over that in an average soil, the total amount applied in 25 years of spray irrigation will still be within the normal range of cadmium found in soils. Recent research by John, et al. (1972) has shown that even 90 pounds of cadmium added to a soil surface over several years did not move more than four inches into the soil profile. Their work on cadmium uptake by oats and studies by Traynor and Knezek (1972) in Michigan on corn show little increase in plant uptake at the levels of cadmium to be applied. Unless very soluble and mobile complexes are formed with low molecular weight organics, there should be little or no movement into drainage.

Lagerwerff (1972) has recently reviewed cadmium in the environment, including soils and plants.

#### Chromium

The amount of chromium which will be added to the soil (Table 9) is well within the amounts normally found in soils. Walter, Traynor, and Knezek (1972) have found that certain soils have a large capacity to fix chromium and leaching into the drainage water should not be a problem. The uptake of chromium by plants has been reviewed by Pratt (1966). Many times when naturally occurring chromium toxicity to plants was suspected, the concentrations of chromium in plant tops was the same as for normal plants (Allaway, 1968; Pratt, 1966). Walter, et al. (1972) found

similar results with corn in Michigan, but Turner and Rust (1971) found uptake of chromium by soybeans from nutrient solution. They did show apparent toxicity to soybeans in soil culture at 10 pounds of chromium per acre, but no uptake data were given for plants grown on soil culture. Apparently no danger exists from injection into the food chain, but the toxicity potential of chromium in the proposed system is not clear. There could be a plant toxicity after approximately five years. Soanne and Saunder (1959) have discussed chromium toxicity from high chromium containing soils.

#### Cobalt

Cobalt contents of most soils range between 2 and 80 pounds per acre, but many values greater than 80 pounds per acre have been reported (Vanselow, 1966). The total cobalt content of soils growing normal forage pastures have been reported to contain up to 80 pounds of cobalt per acre. Use of various chemical extractions to determine available cobalt indicates that the ion is fixed rather rapidly in soils either as an organic complex or on the soil exchange sites. Cobalt addition up to 200 pounds per acre did not cause toxicity to plants in studies by Fujimoto and Sherman (1950). The proposed rate of three pounds per acre per year in the effluent will be rapidly fixed in the soil, will not leach into drainage water and will not significantly increase plant uptake for several years. A soil profile distribution similar to that for cobalt in Figure 5 is expected. Only the addition of highly soluble organic materials will cause significant leaching of the cobalt to the drainage water. Rat feeding trails have shown that animals can tolerate 25 ppm cobalt in plant tissue without adverse affects. In most instances the addition of cobalt to the diet of livestock is recommended for better nutrition (Vanselow, 1966).

### Copper

Most of the copper will be bound in organic or clay complexes near the soil surface. The bonding of copper in soils has been reviewed by Knezek and Ellis (1972). A profile distribution similar to that given in Figure 5 should prevail. There should be no toxicity problems (Reuther and Labanauskas, 1966) and minimal leaching of organic complexes containing copper into the drainage water will occur (Mitchell, 1964). Reuther and Labanauskas (1966) have reviewed the chemistry and toxicity of copper in soils. The soils proposed should adsorb the copper being added for an indefinite period of time at the rate and quantity being proposed if the soil pH is maintained near 7.0.

### Iron

The iron added to the soil is considerable in quantity but will be rapidly fixed by precipitation and surface adsorption (Ellis and Knezek, 1972; Mitchell, 1966). Iron is rarely toxic to plants (Wallihan, 1966). There may be some leaching into the drainage water in the form of organic complexes due to high level of iron addition, but the level should be less than the 0.3 ppm drinking water standard. Addition of such large quantities of iron to the root zone of plants could influence plant nutrition by interaction with the uptake of several nutrients such as manganese and zinc (Mitchell, 1964; Sauchelli, 1969; Wallihan, 1966).

### Lead

Lead addition to the soil will be relatively low. A soil profile distribution pattern similar to copper and cobalt in Figure 5 can be expected. There should be no movement into the drainage water, and plant uptake will probably not be influenced at the levels applied. However, Cox and Rains (1972) have reported considerable lead uptake from severely contaminated soils (64 to



196 lbs of lead per acre). The subject has been reviewed by Brewer (1966) and Lagerwerff (1972).

#### Manganese

When manganese is added in excessive quantities, soluble organic complexes can move through the profiles. In the amounts to be added in the present effluent all of the manganese will be fixed in the profile with no significant quantity going to the drainage water. There is no possible plant toxicity danger if the pH is maintained at pH 7 or above (Mitchell, 1964) and there may be a real possibility of manganese deficiency induced by the high pH and iron application levels (Sauchelli, 1969). The subject of manganese in soil and plant systems has been effectively reviewed by Labanauskas (1966).

#### Mercury

The chemistry of mercury in soils is similar to that of copper and lead. The soluble mercury will be rapidly fixed by the organic and clay fractions of the soil and there will be little movement through the profile and probably no significant increase in plant uptake at the levels being applied in the effluent. Extreme potential toxicity of certain forms of mercury such as methyl mercury, warrant a close investigation of the form being added to the soil and subsequent soil conversions. Lagerwerff (1972) has recently reviewed mercury chemistry and toxicity in plants and soils.

#### Nickel

The chemistry of nickel in the soil is similar to that of cadmium or zinc. The soil should have adequate capacity to adsorb nickel without appreciable leaching to drainage water and without toxicity to plants (Traynor and Knezek, 1972). The high amount of iron being added in the effluent will probably retard

the plant uptake of nickel (Crooke and Knight, 1955). Numerous studies of nickel toxicity from serpentine soils have been made. The levels being added in the effluent of the present system should give no serious problems (Hunter and Vergnano, 1952). The soil and plant chemistry of nickel has been reviewed by Vanselow (1966).

#### Selenium

The knowledge of selenium chemistry in soils is limited. Where selenium additions to low-selenium soils have been followed by repeated measurements of selenium uptake by plants, over 90 percent of the added selenium remains in the soil even after two or three years of continuous cropping and plant removal (Allaway, 1968). Nearly all of the soils on which high selenium (above 5 ppm) grasses and grains have been grown are neutral to alkaline and frequently contain free calcium carbonate and calcium sulfate. The difference in selenium availability in acid and alkaline soils has been attributed to the formation of insoluble compounds or complexes of ferric iron and selenite in acid soils and to the formation of relatively soluble selenates in alkaline soils (Kubota and Allaway, 1972). Kubota and Allaway (1972) classify Michigan as a low selenium area with over 80 percent of all forage and grain containing less than 0.05 ppm selenium. Nevertheless, Hodgson, *et al.* (1971) have shown an area in Michigan, including the proposed spray site, to have forages containing a median concentration of 0.1 ppm selenium. A number of samples contained 0.1 to 0.5 ppm selenium.

Underwood (1962) has quoted work indicating that soils containing more than 0.5 ppm selenium should be regarded as dangerous. Plants have been divided, from the point of view of toxicity, into three classes according to their capacity to assimilate selenium. These are: (1) those showing a limited tolerance (up to about 5 ppm); (2) those which absorb moderate amounts (up to about 30 ppm); and (3) those accumulator plants that absorb more than 30 ppm. Most grasses are in the first class and all cereals are in the second class.

There could be some leaching into the drainage water as some forms of selenium are mobile, some potential for plant uptake from soil at toxic levels for animals and humans and a possibility of plant toxicity after a few years of application. The soil and plant chemistry of selenium has been reviewed by Ganje (1966).

#### Zinc

Although the zinc additions appear to be high, similar amounts of zinc have been added to comparable soils to prevent deficiency on field beans and corn in Michigan (Vinande, *et al.*, 1967). The soil bonding reactions have been reviewed by Ellis and Knezek (1972) and a profile distribution similar to that given in Figure 5 is expected. There should be no leaching to the drainage water and no plant toxicity danger unless a sensitive crop such as field beans is grown on the soil (Melton, *et al.*, 1970). Allaway (1968) has indicated that some increase in the zinc content of crops used for feed would be nutritionally beneficial. The soil and plant chemistry of zinc has been reviewed by Chapman (1966).

#### Special Considerations

The foregoing evaluation has been made on the assumption that the metals in the effluent will be inorganic ions, metal precipitate suspensions or weak natural metal organic complexes. Addition of relatively powerful synthetic chelating ligands or metal chelates of substances such as nitrilotriacetic acid (NTA) or ethylenediaminetetraacetic acid (EDTA) or similar compounds will drastically alter the conclusions of this report. The presence of chelating ligands will increase the mobility of metals through the soil profile, increase plant uptake, reduce soil fixation of metals and either increase or decrease plant toxicity depending upon the nature of the system. Substantial quantities of metals could leach into the drainage water if synthetic chelates are present.

At the organic loads specified in the data provided for this report, it appears that there should not be a significant danger of mobilizing metals through the profile and into the drainage. However, if the organic load is increased and contains 50 percent fulvic acid compounds as indicated for secondary effluent by Rebhun and Manka (1971), significant metals loads could be moved through the soil profile and into the drainage water.

### PART 3 -- CARBON AND NITROGEN REACTIONS IN THE SOIL

#### Internal Cycles of Carbon

The central block of compartments as was shown in Figure 1 represents a broad classification of processes, as well as materials, in the organic phase of soils. Two-directional flows between compartments do not represent reversibility of essential processes. Rather, reversal of flux between compartments is a function of changing oxidation-reduction status of the total system and specific effects of these changes on each of a complex medley of competing processes -- biological and non-biological.

Major cyclic fluctuations in oxidation-reduction status are effected by the soil microflora as they dissipate carbon from input substrates. Dead tissues contain a variety of substrates covering a wide range of energy availability. Changes in the spectrum of available substrates at successive stages in decomposition will bring about successional changes in the composition of microflora populations, their biochemical competencies, and the nature of their metabolic products (Alexander, 1971; Burges, 1967; Clark, 1967; Kononova, 1966).

Although major control of oxidation-reduction status is effected by soil heterotrophic microflora populations, the mass of their tissues is small relative to the total organic phase. In most soils, 80% or more of the organic matter is found in the form of pigmented products of nonenzymatic condensation (humins, humic acids and fulvic acid in Figure 2).

The synthesis of humic substances proceeds by oxidative polycondensations in which numerous exometabolites (notably amino acids, polypeptides, amino sugars, polyuronide residues), and numerous mineral species (notably Fe, Al, Ca,  $\text{NH}_3$ , S) are incorporated as bridging structures between phenolic and polycyclic derivatives of lignin, tannin and numerous plant and microbial pigments (Flaig, 1966; Hurst and Burges, 1965; Kononova, 1966; Stevenson and Butler, 1969). The resulting products are not discrete chemical entities but a system of polymers with



essentially similar constitution, varying continuously over a range of molecular size and chemical and physical properties (Figure 3).

Metal cations are extensively complexed into fulvic acid and humic acid molecules during their formation. Further complexation occurs by adsorption on colloidal silicates and sesquioxides. Susceptibility to fractional extraction is, therefore, a function of molecular size, polarity, and the extent to which bridging metal cations are displaced by ionic or chelate exchange with extracting reagents. Humin is extremely difficult to extract because of its intimate complexation with colloidal minerals. In Figure 1, humin would be considered a humic complex.

Fulvic and humic complexes are much less susceptible to microbial attack than are most substrates present in dead tissues. The half-life of common crop materials decomposing in soil is 1 to 18 months (Clark, 1967; Kuo and Bartholomew, 1966; Sauerbeck, 1968), whereas the half-life of humified residues remaining after 50 to 80% decomposition may be 20 to 50 years (Fredrick and Broadbent, 1966; Jansson, 1971). Under natural vegetative covers, large accumulations of mineral-enriched humic substances have built up over geological periods of time. Half-life estimates (mean residence time for  $^{14}\text{C}$ ) in fulvic and humic extracts from soils range from a few hundred to several thousand years (Paul, *et al.*, 1964; Scharpenseel, *et al.*, 1968). Apparent ages are greater in soils that developed on calcareous mineral strata, notably under grass, than those developed in acidic environments low in divalent basic cations.

In nature, the formation of fulvic and humic complexes is the prime mechanism for stabilizing minerals in the earth's surface mantle against the solvating action of organic acids and chelating agents produced during decomposition of dead tissues. Both the solvating action of metabolites and this stabilizing mechanism are dramatically expressed in podzolic soils where dissolution of minerals gives rise to bleached  $A_2$  horizons and eluviated minerals are deposited as fulvic complexes in the B horizon.

The possibility that humification systems can be manipulated to promote translocation and stabilization in depth through the soil, thereby extending both the capacity and the useful life of waste treatment systems, has been noted (McGauhey and Krone, 1967). Parameters for manipulative control will be discussed later.

#### Cycles of Nitrogen

Of the mineral nutrients, the transformations of nitrogen have been most intensively studied. These transformations, and concepts relating them to organic matter decomposition, soil formation and agricultural practice, have been described in numerous reviews (Bartholomew, 1965; Broadbent and Stevenson, 1966; Delwiche, 1965; Harmsen and Kohlenbrander, 1965; Jansson, 1958, 1971; Mortland and Wolcott, 1965; Patrick and Mahapatra, 1968; Stevenson, 1965b).

Concepts which appear relevant to management of crops and soils for waste treatment purposes will be discussed briefly in connection with Figures 7, 8 and 9. These three figures depict changes in flux of N which might be expected in a sequence of decreasing energy/N ratios in the nutritional environment of the soil heterotrophic population.

Introduction of carbonaceous wastes of low nitrogen content (Figure 7) will lead to rapid net transfer of N from the mineral N pool into the organic phase. The immobilized nitrogen is tightly cycled within the organic phase to meet the requirements of successive populations of heterotrophic organisms.

The size of these populations and the intensity and duration of net immobilization are determined by factors (assumed here to be N) limiting the synthesis of active protoplasm and the rate of respiratory loss of carbon. The period of intense demand and rapid transfer of mineral N coincides with the period during which 15 to 30% of input carbon is being respired. Net transfer into the organic phase continues at a declining rate until about 50% of input carbon has been respired.

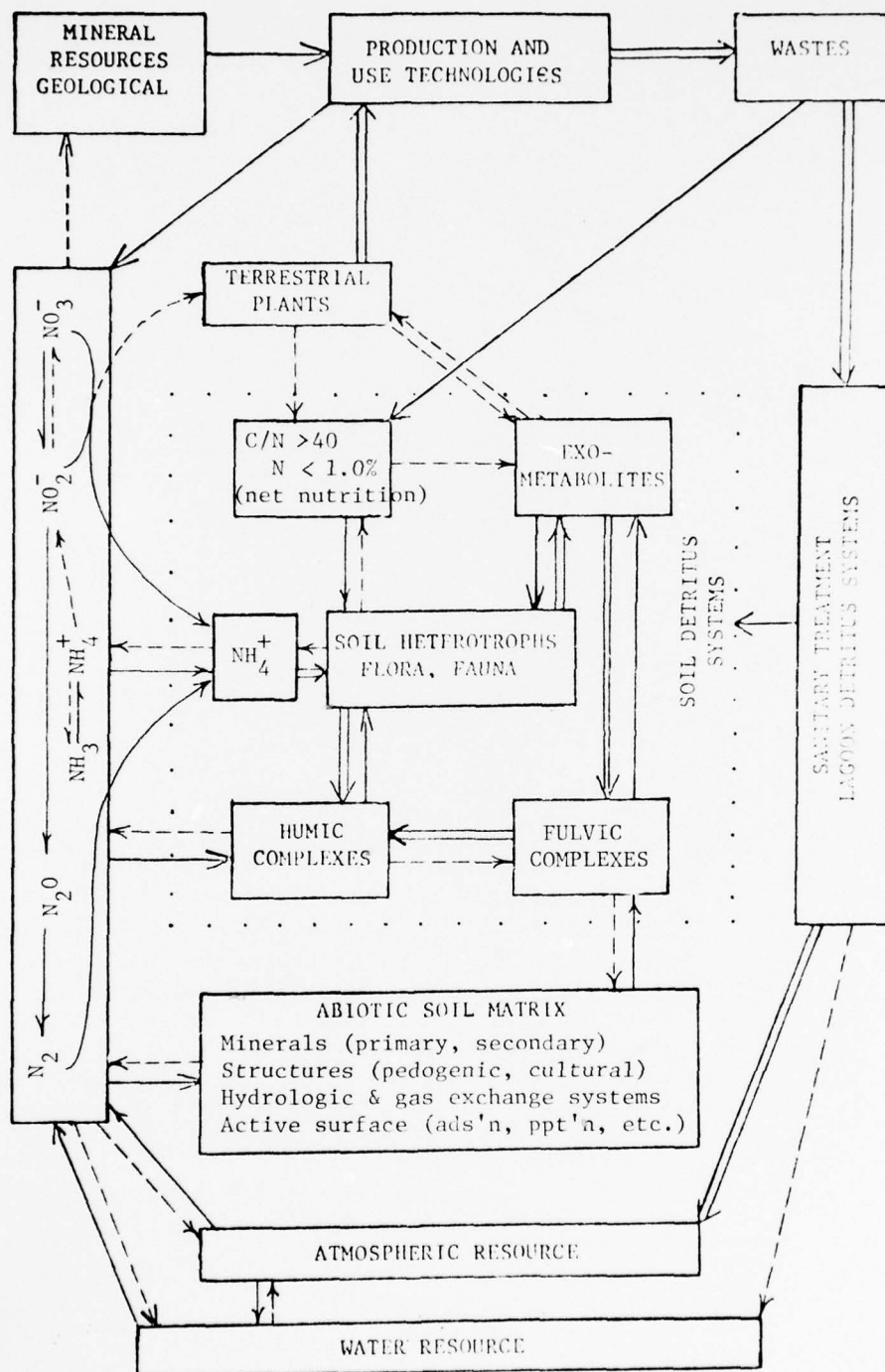


Figure 7. Nitrogen flows with high energy/N ratio:  
 ———> Primary flow  
 ———> Intermediate flow, or net flow  
 - - - -> Restricted flow, or intermittent flow



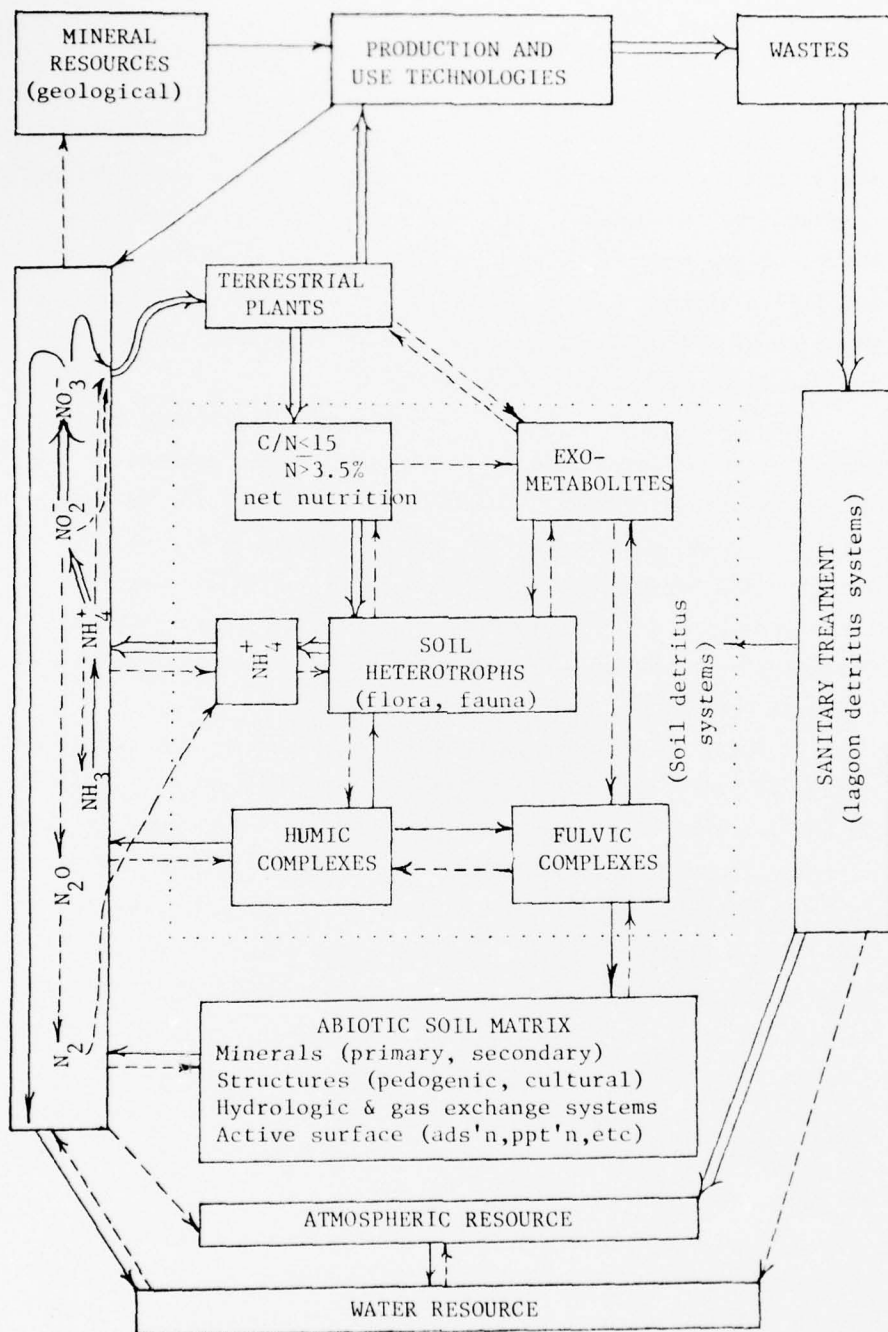


Figure 9. Nitrogen flows with low energy/N ratio:

- ⇒ Primary flow
- Intermediate flow, or net flow
- > Restricted flow, or intermittent flow



At this point, a quantity of nitrogen characteristic for different plant materials will have been stabilized in association with the unrespired carbon. The net nitrogen which has been stabilized includes nitrogen in the added material plus nitrogen removed from the mineral pool, and ranges from 1 to 2 1/2% by weight of the originally added dry material (Bartholomew, 1965). Negligible release of this stabilized N to the mineral pool occurs until about 80% of input carbon has been respired, leaving a residue with a C/N ratio approaching 10.

During the initial period of intense heterotrophic activity, there may be an accelerated attack on soil humic substances (so called "priming" action). However, at the end of the decomposition cycle described above, nitrogen added in the plant material and immobilized from the mineral pool will have been transferred, as a net accretion, to fulvic and humic complexes. Subsequent net release will occur slowly at decomposition rates for humic substances of 1 to 5% per year.

In addition to these pathways of assimilation and complexation, mineral N species are subject to numerous other transformations in the presence of a large energy supply. As diagrammed in Figure 7, these include denitrification ( $\text{NO}_3^- \rightarrow \text{N}_2$ ) and nitrogen fixation ( $\text{N}_2 \rightarrow \text{NH}_4^+$  within the cells of heterotrophic symbiotic and non-symbiotic N-fixing bacteria and blue-green algae). Direct chemisorption of  $\text{NH}_3$  and  $\text{NH}_4$  from the soil solution or the soil atmosphere will exceed any direct exchanges in the reverse direction.

In net effect, a large energy input will cause the soil detritus system to be a sink for mineral nitrogen species, promoting net movement of nitrogen into terrestrial systems from the atmosphere and from the hydrosphere.

The supply of nitrogen for higher plants can be severely restricted by the competing pathways in Figure 7 (plants with N-fixing symbionts are an exception). At intermediate and low levels of energy relative to available sources of N, the primary flow of N (Figures 8, 9) parallels the primary flow of carbon (Figure 2). Nitrification becomes an important reaction in cycles

external to the soil detritus system. Under the conditions of Figure 9, nitrate becomes the principle form of N available to higher plants.

As depicted in Figure 8, an intermediate energy balance tends toward an equilibrium between immobilization and mineralization. The mean annual content of N in the organic phase may vary cyclically but over a narrow range about a relatively constant mean over periods of years.

Such an equilibrium may be approached under climax vegetation where inputs of nutrients from outside the system are small. Under these conditions, the active components of the biomass, (living plants and decomposer populations) are limited in mass to that which can be supported by nutrients cycling within the system (Odum, 1969). In other words, the annual balance between primary production and respiratory loss is determined by the size of the internally cycling pool of one or more critical nutrients.

Such a balance is, in fact, rarely sustained over long periods of time in nature. In the absence of technological influences which differentially favor respiratory loss, net accretion of organic matter and immobilized nutrients has continued over geological periods of time in many natural situations (Bollen, 1959; Mattson & Koutler-Andersson, 1954, 1955; Paul, *et al.*, 1964; Scharpensell, *et al.*, 1968).

The energy balance of Figure 9 represents the situation that existed during the first 60 to 100 years after virgin soils were brought into intensive agricultural production in this country (Stevenson, 1965b). The homeostatic balance between annual production and annual respiration was shifted in the direction of net respiratory loss. Heightened respiration and accelerated succession of decomposer populations led to net mineralization of N and other nutrients which had previously accumulated in humic complexes.

A number of factors contribute to accelerated attack on normally resistant humic substances when virgin soils are brought under cultivation. Tillage disrupts structures which reduce

accessibility of humus to microbial enzymes. Tillage and drainage increase aeration. Lime and fertilizers remove nutritional restrictions on size and activity of decomposer populations. Residues from many cultivated crops are inherently quick to decompose and quantities of residues returned annually are reduced by harvest.

Net mineralization of N and other nutrients from accumulated humus was a principle source of nutrients for crop production during the early years of agriculture in this country. The initial release was rapid. It has proceeded in decelerating fashion, approaching a new equilibrium at soil organic matter levels characteristically 50% lower than in the virgin state.

The greatly increased crop production of today is supported in large measure by direct and recycled increments of nutrients supplied from commercial fertilizers. The situation again approaches that in Figure 8. It differs from the situation under climax vegetation in that there is a greatly increased flux of nutrients paralleling energy flow through plants and heterotrophic populations in the field and, more fundamentally, in the extensive technological translocation and concentration of energy and nutrients in the form of wastes and of mobilized nutrients in treated sewage effluents.

Concentrations of mobile nutrients in waste effluents are a major threat to the water resource. Their control will call for expenditures of technological energy corresponding, if not equivalent, to energy expended in their translocation and concentration, plus biological energy equivalent to that dissipated through BOD reduction by lagoon detritus systems.

Adequate biological energy can be supplied by appropriately managed vegetation in land disposal systems.

#### Probable Impact of Organics and N

##### A. Impact of nitrogen:

Anticipated inputs of organic materials and N will not be great if effluent of the composition in Table 1 is used at hydro-

logically feasible rates. Calculated estimates for carbon and N are shown in Table 11a. At 70 inches per year, total inputs of carbon, organic N and mineral N will be 420, 69 and 54 pounds per acre, respectively. The ratio of carbon to organic N is 6/1. The organics are, therefore, highly enriched with N (approximately 8% N, assuming 50% C).

A substantial portion of this organic N will be quickly mineralized in the soil. The first year's release can be estimated indirectly if it is assumed: (1) BOD will be further reduced by 80%, (2) the more resistant materials (COD minus BOD) will be reduced 10%, and (3) the residue will have a C/N ratio of 10. The calculated release for 70 inches per year is 37 pounds per acre. This added to 54 pounds ( $\text{NH}_3 + \text{NO}_3^-$ ) in the effluent gives 91 pounds of mineral N available for crop use. An additional 60 to 80 pounds of fertilizer N would be needed for a 150-bu corn crop which would be feasible on these soils under irrigation and good management.

In Table 11b, parallel calculations have been made for 70 inches per year, using the effluent profile in Table 2-1 of the CRREL Report (1972). This may represent a more typical effluent, with 2 1/2-fold more total N than that shown in Table 11a. The N is 90% mineralized and extensively nitrified (2 ppm organic N, 9.8 ppm  $\text{NH}_4\text{-N}$ , 8.2 ppm  $\text{NO}_3\text{-N}$ ). The net increment of mineral N to the end of the first year is 291 lb/a, or 3 1/2 times greater than in Table 11.

Under management systems proposed in the preliminary Dow Report (June, 1972), the critical point in the rotation for removal of this quantity of N will be when corn is grown.

This quantity of N (291 pounds) is approximately what would be contained in grain plus stover plus roots and stubble of a 156-bushel corn crop ( $146 + 97 + 48 = 291$  pounds, assuming that N content of grain is 1.7%, and that the quantity of N in stover is 2/3 of that in grain and N in roots and stubble is 1/3).

Part of the nitrogen in this crop must be credited to nitrogen

Table 11a. Estimated annual inputs and conversions in soil of carbon and nitrogen in secondary effluent.

Component	Concentrations in Effluent		Annual Increments											
	mg/l lb/a/in		52 wks./yr.				52 wks./yr.				52 wks./yr.			
	0.5	1	2	4	in/wk	0.5	1	2	4	in/wk	0.5	1	2	4
Effluent (in/yr)														
A. Inputs:														
(1) COD	70													
(2) COD carbon <sup>1</sup>	26	6.0												
(3) BOD	13													
(4) BOD carbon <sup>1</sup>	5	1.2												
(5) Resistant carbon <sup>2</sup>	21	4.8												
(6) Organic N	4.4	.99												
(7) Mineral N (NH <sub>3</sub> + NO <sub>3</sub> )	3.4	.77												
B. Conversions first year:														
(8) Loss of BOD carbon (80%)	17	34	67	134										
(9) Loss of resistant carbon (10%)	8	17	34	67										
(10) Total carbon loss (8) + (9)	25	51	101	201										
(11) N mineralized (6) - (13)	9	19	37	74										
C. Net increments to end of first year:														
(12) Organic carbon (2) - (10)	80	160	320	640										
(13) Organic N <sup>3</sup> (12) x 0.1	8	16	32	64										
(14) Mineral N (7) + (11)	22	46	91	182										

<sup>1</sup> COD x 12/32; BOD x 12/32<sup>2</sup> (2) - (4)<sup>3</sup> Assuming residual C/N = 10



Table 11b. Estimated annual inputs and conversions in soil of carbon and nitrogen in secondary effluent, using effluent profile in Table 2-1, p. 3, CRREL Sp. Rep. 171, May 1972, and annual applications of 70 inches (2 in/wk for 35 wks).

Component		Concentrations in effluent		Annual increments
		mg/l	lb/a/in	at 70 in/yr.
				lb/a
A. Inputs:				
(1)	COD	70		
(2)	COD carbon <sup>1</sup>	26	6	420
(3)	BOD	25		
(4)	BOD carbon <sup>1</sup>	9.4	2.1	147
(5)	Resistant carbon <sup>2</sup>		3.9	273
(6)	Organic N	2	.45	32
(7)	Mineral N (NH <sub>4</sub> + NO <sub>3</sub> )	18	4.1	287
B. Conversions first year:				
(8)	Loss of BOD carbon (80%)			117
(9)	Loss of resistant carbon (10%)			27

Table 11b (con't.)

(10) Total carbon loss (8) + (9)	144
(11) N mineralized (6) - (13)	4

C. Net increments to end of first year:

(12) Organic carbon (2) - (10)	276
(13) Organic N <sup>3</sup> (12) x 0.1	28
(14) Mineral N (7) + (11)	291

<sup>1</sup>COD x 12/32; BOD x 12/32

<sup>2</sup>(2)-(4)

<sup>3</sup>Assuming residual C/N = 10

carried over from the previous season. Nitrogen would have carried over principally in two forms: (1) unused mineral N and (2) N in readily mineralizable organic components of roots and other residues from the preceding crop.

In the proposed rotations, the crop preceding corn would have been alfalfa-brome. An estimate of mineral N not used by alfalfa-brome can be approached from Penn State data (Parizek, *et al.*, 1967) which showed 190 pounds N harvested in alfalfa hay. This figure also corresponds to reported recoveries in forage grasses of 70% of fertilizer N applied at high rates (Stanford, 1966).

It may be assumed that N equivalent to about one-half of the harvested N would have carried over in stubble and roots of the alfalfa-brome, or about 90 pounds (Millar, 1955, p. 301). This would leave 11 pounds of unused mineral N to carry over (assuming no leaching loss or denitrification).

In the Michigan Fertilizer Recommendations Bulletin E-550 (1970), mineralization release to the first crop following an alfalfa sod is estimated at 60 pounds N. This plus 11 pounds direct carryover represents 71 pounds mineral N in excess of the above calculated incorporation into a 156-bus corn crop.

It can be assumed that the excess N would nitrify and be subject to leaching and denitrification. Under high moisture conditions and in the presence decomposing crop residues and an active rhizosphere flora associated with a growing crop, denitrification losses can be estimated, conservatively, at 30% (Allison, 1965). Thirty percent of 71 is 21 pounds N/a, leaving 50 pounds N that might leach in 80" net percolate (70" irrigation plus 35" precipitation minus 25% evapotranspiration). The average percolate concentration would be 2.7 ppm N, mainly as  $\text{NO}_3$ .

Corn and forage yields necessary to achieve this degree of N removal can be expected on these soils with appropriate management. High yields are essential. For example, the N in a 100-bus corn crop, calculated as above, would total 191 pounds. The excess current input (100 pounds) plus 71 pounds carryover from alfalfa-brome, discounted 30% for denitrification loss, would calculate

to a percolate concentration of 6.5 ppm N.

With regard to forage mixtures, the legume is the critical component with respect to nutritional value and palatability but, also, from the standpoint of harvest removal of nitrogen and other nutrients. Yields of grass-legume mixtures are frequently 2 to 4 times the yields of the grass in pure stand. (Ahlgren, 1956). Where N is limiting, this is a response to N fixed by the legume. In N-rich environments, N-fixation is inhibited (Vincent, 1965) but the nature of the plant still leads to larger production and a higher content of N and other minerals in the legume component of grass-legume mixtures under conditions favorable to both.

A critical consideration is the fact that legumes are much less tolerant of excessively wet situations than are most of the grasses considered. Timothy and reed canarygrass can withstand prolonged periods of waterlogging, even flooding, whereas even the most tolerant legumes (alsike, ladino clover) cannot. Disease problems with legumes and brome grass are also intensified under humid conditions.

These differences in water tolerance imply that irrigation rates will be varied over the rotation, a concept which has favorable theoretical implications noted in the next section.

#### B. Impact of organics:

No significant short term effects of organics on important soil properties at hydrologically feasible rates are anticipated. Long term effects cannot be predicted. However, the nature of organics identified in secondary effluents have implications which should be kept in mind.

Analysis of secondary effluents in Israel showed 40 to 50% to total COD in "humic substances" of which the fulvic fraction constituted 50 to 75% (Rebhun and Manka, 1971). Carbohydrate and proteinaceous residues comprised 11-12% and 21-24%, respectively, of the total COD. Anionic detergents and ether solubles comprised most of the remainder (12-15% and 6-10%, respectively), but a significant 1 1/2-2% tannins was also found.

Extraction of membrane ultrafiltrates (100 MW cut-off) with polar and nonpolar solvents reveals the presence in secondary effluents of numerous monomeric species, prominent among which are aliphatic and aromatic acids, amines and amides (plus chlorinated phenols and alkylphenyl compounds of probable industrial origin) (Arthur D. Little, Inc., 1971).

The high content of highly acidic fulvic materials (Figure 3) and the presence of substantial proportions of aromatic polymers (tannins) and monomeric species with strong detergent and chelating properties have profound implications for the fate of metals input with municipal effluents, as well as for the integrity of minerals in the abiotic soil matrix. Monomeric chelates and organo-mineral complexes formed in lagoons and in the soil can be expected to display a degree of mobility in percolating water. The initial mobility, in particular, will be great because of the high enrichment with N and because of reducing conditions associated with high moisture in the receiving soils.

It is expected that most, if not all, of these mobile materials will be stabilized by complexation with soil minerals and humic materials. The extent to which this happens, however, will be influenced by management practices.

#### Management for Stabilization of Mobile Waste Components

Factors which influence mobility of organics and minerals in soils and which can be controlled to a degree by management include:

(1) Oxidation-reduction:

In general, reducing conditions promote mobility of organics, transition metals and associated anions such as  $\text{PO}_4$ . Oxidation promotes organic condensations, organo-mineral complexation and precipitation of mineral ions and oxides.

(2) Soil pH:

Organic condensations, and mineral complexation and precipitation reactions are, in general, promoted by increasing pH over the acid to low alkaline range. Periodic liming to maintain optimum



pH for crops (pH 6.5-6.8) would promote the desired reactions, as well as the desired high level of crop production.

(3) Cycles of wetting and drying:

Drying results in shrinkage, cracking, and restoration of permeable structure to massive impervious clay strata. Deposition of organics on fracture planes in such strata during cycles of wetting and drying reduces their cohesiveness and their tendency to seal off in succeeding cycles of wetting.

(4) Reaction time:

Many of the reactions leading to stable equilibria in soils have a time course of weeks, months or years. The use of resting periods to promote aerobic recovery and drying and to extend reaction time is an accepted principle in management of soil systems for wastewater treatment (McGauhey and Krone, 1967).

(5) Crop residue return:

Condensation reactions leading to the formation of stable humic complexes take place during the course of decomposition of fresh plant residues, rather than during a later sequence of reactions among already oxidized products. A continuing supply of less highly oxidized carbon skeletons will be essential for stabilizing the low energy organics in secondary effluents. Humification reactions can contribute importantly to inactivation of potentially toxic metals and non-metals.

Under management programs proposed in the preliminary Dow Report (June, 1972), forage species will occupy the irrigated areas up to 80% of the time. The grass components in the proposed forage will supply large quantities of unharvested residues.

Production of grass stubble, roots and rhizomes in the first season normally equals the dry matter in shoots and leaves. The proportion of roots and rhizomes normally increases in succeeding years (Ahlgren, 1956; Russell, 1961, p. 578). High N fertility tends to reverse this trend, leading to greater production of tops (Watkins, 1940). However, high moisture status favors the production of underground parts (Gardner, 1942). Up to three tons of roots and corms have been reported under fertilized timothy

(Millar, 1955, p. 302). Larger values might be expected under a vigorous rhizome producer such as reed canarygrass.

Increases in soil organic matter of 1200 to 1800 lbs/acre/year (60 to 90 lb N) have been observed where previously cultivated soils have been retired to permanent grass cover (Russell, 1961, p. 318). In these studies, hay was not harvested but species of relatively low productivity were involved. It was predicted that these rates of organic matter accumulation, observed over a 25-year period, would decline, tending to a new equilibrium after 100 years similar to that found in old pastures in the area.

Leveling off of net production and total biomass is characteristic of closed systems restricted to nutrient supplies within the system. Where nutrients are imported into the system from outside, net incrementation of organic mass can continue indefinitely at rates determined by annual inputs of minerals, the productive capacity of vegetation and the respiratory capacity of detritus systems (Odum, 1969; Pomeroy, 1970).

In lake-bog-moor sequences, annual net incrementation of organic matter has been sustained over geological periods of time. In a Swedish bog studied by Mattson and Koutler-Andersson (1954, 1955), the only sources of nutrients were in rainwater and wind-blown dust, and the vegetation was sphagnum. Based on pollen chronology, the average rate of accumulation over the period 1,000 A.D. to 1942 was 0.74 mm or 1,040 pounds peat per acre per year. Accumulation of minerals increased abruptly after 1500 A.D., coincident with increased human occupancy and industrial activity in northern Europe. Incorporation of N increased from 6 to 16 pounds and ash constituents from 7 to 37 pounds per acre per year. Ash constituents included oxides of Si, Al, Fe, Mn and Ti.

Under the cropping programs proposed in the Dow Report, it can be anticipated that net accumulation of residual organic matter will occur, even if above ground production is harvested. If a high level of production is maintained, annual increments of residual humus can be estimated, conservatively, at 1,000 pounds per acre, more likely double that figure. This can be a very important

factor in inactivation of potentially toxic minerals. It will also reduce, substantially, estimates of nitrogen which may enter drainage. In addition to immobilization, microflora associated with grass roots are characterized by a high level of denitrifying activity (Woldendorp, 1962). Nitrate concentrations under grass vegetation are normally of the order of 1 ppm or less (Russell, 1961, p. 309; Clark and Beard, 1960).

To maximize removal of nutrients in harvested crops and by immobilization (and denitrification in the case of N), it will be important to maintain high yields of all crops. The factor most likely to limit yields will be excess water (see Part 4). On most of the soils considered, 70 inches per year will almost certainly be excessive for legumes, brome grass and corn. Timothy and reed canarygrass might accept water at this rate if advantage is taken of the fact that they can withstand periodic flooding during winter and early spring when they are dormant.

The limiting hydraulic property of these soils is the low permeability of the subsoil when maintained continuously in a wet condition. The crops considered lend themselves to rotational management of irrigation inputs in patterns which can lead to improvements in permeability over time.

Reducing conditions during wet portions of the rotation under water-tolerant grasses would promote movement of mobile organics and minerals in depth through the soil. At the other end of the rotation, lower water inputs of alfalfa-brome would improve aeration and promote deep root penetration. The third year of alfalfa-brome might, in fact, be left without irrigation to promote deep de-watering and restoration of permeability in the subsoil.

Aeration and de-watering during this drying sequence would promote stabilization of translocated materials by oxidation, irreversible drying, chemical precipitation and reversion to slowly soluble forms. The interaction of illuviated organics with subsoil clays could tend to reduce their plasticity and lead to permanent improvements in permeability over time.

As a practical point, it may be necessary to approach clean cultivation in the second crop of corn to facilitate preparation of a suitable seedbed for reed canary. This grass is an aggressive competitor once it has established itself. However, the seeds germinate slowly and the grass is non-competitive in its seedling stages (Ahlgren, 1956; Whyte, et al., 1959). Perhaps the reed canary could be seeded to wide rows (4 feet or wider at the same time with timothy and clover sown in between). The timothy and clover would provide early production and be superceded by reed canary over a period of years. This approach would also eliminate one problematical seeding operation.

#### PART 4 -- SOIL PHYSICAL PROPERTIES AND THEIR LIMITATIONS

The physical properties of a soil govern the rate at which the soil will receive or infiltrate water and the rate at which the water is passed through the soil. These same physical properties control the depth to which a given quantity of water will penetrate and govern the air-water relations of the soil.

Physically the soil is made up of the inorganic and organic particles which form the soil matrix. The soil water or liquid of the soil is associated with this soil matrix. The remainder of the soil volume is occupied by the soil air or gaseous phase. The solid matrix determines the total pore space. The amount of water on the matrix which in wet soils is held primarily by surface tension in turn determines the amount of air filled pore spaces. The continuity and size of these pores influence the soil air relations and greatly influence the infiltration of water into the soil at the surface (infiltration rate) and the permeability of soil (the rate at which water passes through a particular section of soil).

For wastewater disposal the most important physical properties are those of infiltration rate and permeability (McGauhey and Krone, 1967). The lowest of these rates will determine the rate at which wastewater can be successfully applied.

The soils of concern in this project are loamy soils having a mixture of sand (2 to 0.05 mm), silt (0.05 to 0.002 mm) and clay (less than 0.002 mm) material in them. This gives a matrix which has finer pores and correspondingly lower infiltration rates and permeabilities than sand or silt soils. The fine particles in these soils, nevertheless, make them better for wastewater renovations.

Loamy soils can have higher infiltration and permeability rates than would be expected from their dense matrix of sand, silt and clay because of the phenomena of soil structure formation. This is a means by which the individual particles clump together usually under the influence of organic matter and certain inorganic



polycations to form aggregates. Thus the soil matrix becomes a bimodal system of a fine matrix inside the aggregates and coarser matrix between the aggregates. A well aggregated soil can therefore have properties of both the fine textured soil and the coarse textured soil. The coarse matrix allows faster water movement through the soil and fine matrix is available to interact with the waste contaminants if the application rate is not too rapid.

The difficulty with soil structure is that it is both fragile and dynamic. It can easily be destroyed by excessive wetness, manipulation (particularly when wet) and a whole series of natural and artificial processes. The structure is formed under the influence of plant roots, drying, freezing and thawing, etc. Baver (1956) provides a rather complete discussion of these phenomena. Soils will be better for wastewater disposal and renovation if their structure is preserved and improved. The structure of the surface has a tremendous effect on infiltration and the structure of the subsoil influences permeability.

The measurement of meaningful infiltration rates and permeabilities is difficult (Parr and Bertrand, 1960) and it is also complicated by the dynamic nature of these properties. We have data on infiltration rates and hydraulic conductivities of Michigan soils of the same or similar soil types as the ones found in the proposed disposal areas. These data and their interpretations are given in Table 12. The infiltration data was taken with double ring infiltrometers (5" & 9" rings). "Where data does not exist estimates are given as to the relative permeabilities according to the following:

<u>Rate classes</u>	<u>Rate of inches per hour</u>
Very slow	Less than 0.20
Slow	0.20 to 0.80
Moderate	0.80 to 2.50
Rapid	2.50 to 10.00
Very rapid	Greater than 10.00

Table 12. Soil limitations for disposal of municipal wastewaters - modified table from Shneider and Erickson, 1972.

Soil Type Group	Soil Series	General Soil Profile	Texture	Natural Drainage	Water <sup>1</sup> Table Range	Permeability <sup>2</sup> (in/hour)	Water <sup>3</sup> Holding Capacity (in/foot)	Slope (Percent)	Remarks	Hydrologic-Limitations <sup>4</sup> Irrigated Surface Sprinkler Surface	Tile Drainage <sup>5</sup> Surface
2.5 b	BLUNT	Surface	Loam	Somewhat Poorly	24-120"	Very Slow	High	0-2	Very Slow Per- meability Seasonal High Water Table	Very Severe	Very Severe
		Subsoil	Silty Clay Loam to Clay			0.2				Very Severe	Very Severe
		Parent Material	Clay Loam to Silty Clay Loam			0.1					
2.5 c	CONOVER	Surface	Loam	Somewhat Poorly	24-120"	Slow	High	2-6	Slow Permeability Seasonal High Water Table	Severe	Severe
		Subsoil	Clay Loam			0.3				Severe	Severe
		Parent Material	Loam to Silt Loam			0.7				Severe	Severe
2.5 d	BROOKSTON	Surface	Loam	Poorly	0-24"	Slow	High	0-2	Slow Permeability Seasonal High Water Table	Very Severe	Very Severe
		Subsoil	Clay Loam			0.8				Very Severe	Very Severe
		Parent Material	Loam to Silt Loam			0.6					
2.5 b	KIBBIE	Surface	Fine Sandy Loam	Somewhat Poorly	24-120"	Slow	High	0-2	Stratified Soils Permeability Seasonal High Water Table	Severe	Severe
		Subsoil	Silt Loam to Silty Clay Loam			0.6				Severe	Very Severe
		Parent Material	Very Fine Sandy and Silty								
3 b	LOXIE			Somewhat Poorly	24-120"	Moderate	Low		Seasonal High Water Table		



4 c	Surface	0-10"	Sandy Loam	Poorly	0-24"	Moderate	Low	High Water Table	Severe	Severe	Moderate	Moderate
	Subsoil	10-18"	Sandy Clay Loam									
	Underlying Material	18-60"	Sand and Gravel									
4/2 b	Surface	0-9"	Loam	Somewhat Poorly	24-120"	Moderate	Medium	Seasonal High Water Table	Severe	Severe	Severe	Severe
	Subsoil	9-18"	Sandy Clay Loam									
	Underlying Material	18-60"	Gravel and Sand									
4/2 c	Surface	0-9"	Loamy Sand	Poorly	0-24"	Moderate	Medium	High Water Table	Severe	Severe	Severe	Severe
	Subsoil	9-32"	Sand to Loamy Sand									
	Underlying Material	32-60"	Loam to Clay Loam									
4/2 c	Surface	0-7"	Loamy Sand	Poorly	0-24"	Moderate	Medium	High Water Table	Severe	Severe	Severe	Severe
	Subsoil	7-28"	Sand									
	Underlying Material	28-60"	Loam to Clay Loam									

#### Other Soils<sup>6</sup>

<sup>1</sup> Natural drainage

<sup>2</sup> Very slow = less than 0.2, Slow = 0.2 to 0.8, Moderate = 0.8 to 2.5, Rapid = 2.5 to 10.0

<sup>3</sup> Inches/5 feet, Low = 10 to 13, Medium = 13 to 18, High = 18 to 23

<sup>4</sup> Moderate = 1/2"/hr. maximum 4"/week, Severe = up to 1/4"/hr. maximum 2"/week, Very severe = none unless physical manipulations as drainage are used

<sup>5</sup> Assume adequate tile drainage

<sup>6</sup> Soils with similar limitation for this study: Parkhill, Tappan and Brookston; Cape, Belding, London and Conover. Peano is more limited than Brookston

Hydrologic limitations are given as four degrees of limitation as follows:

- Slight limitations - up to 1 inch per hour with a maximum of 8 inches per week
- Moderate limitations - up to 1/2 inch per hour with a maximum of 4 inches per week
- Severe limitations - up to 1/4 inch per hour with a maximum of 2 inches per week
- Very severe limitations - impractical to use."

Data obtained with rainfall simulators of both the FA (Rowe, 1940 and Wilm, 1941) and Purdue (Bertrand and Parr, 1961) types are given in Table 13. These data show the differences in infiltration rates that are measured by different methods but also the differences in the same soil under different conditions of the surface and plant cover.

These soils are all very susceptible to slaking of the surface under the impact of water drops from rain or sprinkler irrigation greatly reducing infiltration rates. If a crop cover is used to break the impact of the drops, crusting can be reduced but all row crops have a period when the surface is bare.

Table 13. Collection of rainfall simulator data, wet run.\*

Infiltration - inches per hour.

Soils	Sims Sandy Clay Loam	Miami Loam
Crop		
Corn Seed Bed	1.1	1.2
Corn (knee high)	2.2	1.0
Brome Grass (July)	2.5	1.3
Corn (after harvest)	0.6	
Corn (knee high)**	1.4	

\*Wetted after one hour the day before at 4"/hr.

\*\*FA Infiltrometer (Rowe, 1940) all other data from the Purdue Infiltrometer (Bertrand and Parr, 1961).

One of the processes for soil structure formation that is essential for good infiltration and permeability in loamy soils is that of dehydration. Under the proposed wastewater application



rate of 2 inches per week plus rainfall for 35 weeks these soils will never have a chance to dry and crack to improve and stabilize the structure and increase water conduction.

Cover crops can protect the surface soil structure and maintain the infiltration rates but some crops, such as perenial forage species, may be difficult to harvest or market. Corn, which would be easy to harvest and market, would leave the soil surface bare for a period each Spring until the crop again covers the soil. This will occur around July 1. (A grass winter cover crop with the grass killed by herbicides and the corn planted without plowing might be a solution to the cover crop question although extensive use of herbicides may have adverse secondary effects on non-target biota.) The data reported in Tables 12 and 13 also suffer from the limitation for this application in that these soils were partially dried and then rewetted and the measurements subsequently made on the rewetted soil. This procedure is satisfactory in irrigation practice or for evaluating the effects of rainfall during most of the season. But it does not really measure the case where the soils come out of the wet winter conditions and are kept continually moist by rainfall plus 2" per week of waste application. The limited information we have on this phenomena were taken on the subsoil of a tile drained Sims sandy clay loam soil. The data\* are as follows:

<u>Date</u>	<u>Subsoil Infiltration Rate</u>
July	0.9"/hr.
April	0.03"/hr.

These data illustrate that the clays in these soils hydrate during the winter and under these conditions the structure of the subsoil and deeper horizons becomes massive and has a low permeability. Once the soil dries, as it would naturally (especially under the influence of plant roots), natural structural cracks develop.

\* (From Mich. Agr. Exp. Station Proj. 413)

These cracks seem to be partially irreversible dehydrated which usually allows them to persist until the next winter. Continuous wastewater additions starting early enough to allow for 35 weeks of application will prevent this natural drying phenomena from occurring. Therefore, the permeability measurements and drainage observations made on soils from the usual agricultural or irrigation practice probably do not apply and should be reduced for wet soils that will exist with continuous wastewater application.

Monovalent ions can also affect permeability. McGauhey and Krone, (1967) refer to Lunt as saying that if the sodium adsorption ratio (SAR) is less than 3 adverse long term effects will be avoided but that SAR values between 3 and 5 may cause reduced permeability. The SAR for the Southeastern Michigan soils was calculated to be 2.95. The Detroit effluent values therefore are borderline and certainly would become a problem if, for any reason, the Na or K concentrations would increase over those indicated earlier (Table 2). The continuous application of the effluent would drive the ESP (Exchangeable Sodium Percentage) of these soils as predicted from Figure 22 of Richards (1954) to 3%, or from Figure 23 to 6%. The ratio of air permeability to water permeability (a measure of soil structure stability) of some soils would change greatly at these ESPs. Figures 22 and 23 of McGauhey and Krone (1967) show that these values are on the borderline of causing swelling and reducing hydraulic conductivity. Studies of the Michigan soils is necessary to ascertain their actual behavior. It would be well to also study the effects of a possible increase in the sodium content of the effluent. There is an interaction between soils with appreciable Na on their exchange complex and water without salts, such as the water added from snow melt and early spring rains in Michigan. This water could cause a dispersion of the soil and a reduction in infiltration and permeability. The magnitude of this effect should be studied on these soils.

Most of the soils proposed for the Southeastern Michigan study are naturally imperfectly or poorly drained. This would indicate that the natural water table is probably shallow and in

the winter and spring could be at the surface. Thus these soils must be artificially drained. The Dow Engineering Report (1972) considers tile underdrainage at 33 or 55 feet between centers. If these are placed deep enough this would be quite adequate for agricultural production under normal rainfall conditions where the soil usually dries out to a considerable depth every season under the influence of plants. Deep rooted plants are known to improve the drainage characteristics of these soils probably due to the partially irreversible dehydration of the clays in these soils which causes the development of a relatively stable structure. Under the conditions of adding 70 inches of wastewater per year when there is usually, at most, only a six to twelve inch water deficit during the cropping season, these soils will always be wet. Drying will only occur at the surface, and in most of the soils, would not extend below 18 inches. This drying would not affect the subsoil of these soils which are usually the *finest texture and* least permeable. The low permeability of these subsoils could cause problems in the operation of the system in times of rainfall and low evapotranspiration.

The recommended water application rates on the finer of these soils is only 0.3 inches per hour for irrigation, assuming an application on a reasonably dry surface. The amounts determined by rainfall simulators is of the order of one inch per hour. The rates of application at the ends of the proposed center pivot sprinklers is 2.26"/hr on 100 foot wide spray at 1450 feet from pivot (Dow Report, 1972 Table IV-2) for a 3" application. These amounts would be much too high. A land cell system is proposed to keep the water from moving more than 100 feet. This would cause ponding of the surface which would have an adverse effect on both the soil and the crop. The ponding could cause severe dispersion and loss of infiltration capacity if it occurred every day five days a week, as well as severe crusting when allowed to dry for essential cropping operations. During ponding the crop roots would suffocate and under other environmental stress could reduce crop yields.

With 70 inches of water application during the non-winter season, these soils will always be very wet, and the farming of such soils under normal circumstances often becomes a problem during the spring and fall. Therefore, it may be impossible to perform normal farming operations of seedbed preparation, planting and harvest. It may be possible, however, to develop a system that would eliminate seedbed preparation by no till planting and corn, for example, might be harvested after the ground freezes. This system would warrant further testing.

Any manipulation or traffic on these soils, while the soils are moist, will degrade their structure and reduce the infiltration and permeabilities by an order of magnitude. Tillage manipulations to improve the structure all require a relatively dry soil, which would require the cessation of wastewater application for a period of time during which good drying conditions existed. Deep tillage to improve subsoil conditions could require as much as two weeks to a month in the summer even if the soil had a transpiring crop.

Many of the crops on the fine textured soils of Michigan suffer from lower yields because of poor soil aeration (Erickson and Van Doren, 1960). This condition was found to limit crop yields 40% of the time. The yield reduction occurred during years of excess rain, or when several intense rains occurred during the season. The crop plant roots suffocate during the periods when the pores are largely filled with water because oxygen exchange with the atmosphere is limited. If the suffocation does not kill the plant, as it will under prolonged ponding on days of high radiation and temperature, it will kill some of the plants roots. This root pruning will cause a reduction in yield. The maintenance of a good structure in the surface soil and in the soil throughout the root zone will reduce these effects under normal rainfall conditions by providing fast drainage of excess water from the root zone. However, with two inches per week additional water, these soils will be very susceptible to aeration difficulties which could reduce the yields to zero in

a severe rainy period.

There is a difference in crop susceptibility to poor aeration damage. Grasses and corn are most resistant but sugar beets, potatoes, tomatoes are very susceptible and should be avoided.

Under continued poor aeration conditions organic and inorganic materials in the soil become reduced and the oxidation-reduction potentials of these soils drop and even become negative. The poorly drained and imperfectly drained soils, such as are considered in this study, undergo annual changes of redox potentials in the natural state and to a lesser degree in the drained condition (McKenzie and Erickson, 1954). During the winter the potentials become low, then rise as the soil dries reaching a maximum in the summer and fall. These changes occur first at the surface and then proceed downward. In the reduced condition, these soils may release substances which are toxic to plants. Under these conditions the soils may also lose some of their ability to remove phosphorus and other substances. With large additions of wastewater, it is possible that the redox potentials will recover more slowly, and in some instances, may not recover at all.

There are several manipulations that are used to improve the rate of water removal by drainage tile in agricultural practice. These should be avoided as they would short circuit wastewater soil contact. One such procedure would be to back fill the trench over the tile with more porous material. Another would be to allow the soil to dry and crack in the lower soil horizons so that cracks would lead directly from the surface to tile. Both of these procedures would allow drainage without complete soil contact and would cause failure of the water renovation processes.



## SUMMARY AND CONCLUSIONS

Available information sources on the chemical and physical status of soil types, in areas proposed for spray irrigation of wastewater from the Southeast Michigan, were collected and analyzed. Data on the anticipated quality of the wastewater to be applied to the soil was used in conjunction with the engineering design provided by the Corps of Engineers as the basis for the assessments of this report.

### Phosphorus Adsorption

Estimation of phosphorus adsorbing capacities of soils by Langmuir adsorption isotherms is expected to give conservative estimates. Langmuir adsorption isotherm data for phosphorus adsorption for the major soil types in the area would indicate that these soils are generally considered to be medium in their adsorbing capacity. One, the Parkhill, is medium-low in phosphorus adsorption capacity. A table was constructed to estimate the life of the system in regard to adsorption of phosphorus based on phosphorus adsorbing capacity, inches of effluent applied per week, the number of weeks irrigated per year, and the depth to saturated conditions in the soil. Based on this table spray rates of greater than two inches per week would lead to a very short life of the system as far as effectively removing phosphate. At application rates of 2" per week for 35 wks/yr the phosphorus adsorbing function of the soils will be maintained for approximately 30 years. During this period the water leaving the tile drainage system will be of very good quality with respect to phosphorus concentrations. The expected concentration of phosphorus here would be approximately 0.05 mg/liter.

### Sodium, Potassium and Chloride

The exchangeable sodium percentage and the exchangeable sodium + potassium percentage was calculated for the effluent and for

increasing levels of sodium in the effluent and for increasing levels of sodium in the effluent. It was concluded that the salt content would not restrict most crops grown in Michigan. The exchangeable sodium will increase to a point that may borderline on dispersion problems in the soil. Potassium levels are not going to be a problem.

Although large quantities of chloride are being applied, it is not adsorbed and concentrated in the soil. The levels being applied should not restrict crop growth. However, the soil is not expected to reduce the level of chloride in the water.

#### Nitrogen and Carbon

The probable impact of nitrogen in secondary effluents was assessed, using compartmentalized diagrams relating cycles of nitrogen to internal cycles of carbon in the soil organic phase. The probable impact of wastewater organics on important soil properties was also assessed. Proposed management parameters were evaluated in their relation to maximum harvest removal and stabilization of nutrients in residual humus and dissipation of surplus nitrate by denitrification.

The humification process in detritus systems of soils, as well as lagoons, was shown to have two distinct phases: (1) a biological phase in which  $\text{CO}_2$  is disengaged from fixed carbon compounds, and (2) a non-biological phase in which energy-rich products of heterotrophic degradation and metabolism are condensed, non-enzymatically, to form fulvic and humic complexes of low energy availability.

The distinction between these two phases is important because the essential processes leading to stabilization of organics and immobilization of minerals in soils are non-biological. The organo-mineral complexes formed are durable aggregating agents, giving stability to soil structures essential for water movement and gas exchange.

The degree of stabilization (immobilization) increases in the order: exometabolites < fulvic complexes < humic complexes. The degree of oxidation, however, reaches a maximum in relatively mobile fulvic materials. Thus, a continuing direct input, into the humic acid fraction, of less highly oxidized carbon skeletons is essential for maintaining stability over time. These carbon skeletons are generated by activities of heterotrophic organisms with access to substrates of high energy availability.

Organics in secondary effluent will supply negligible energy to support these essential heterotrophic activities in receiving soil.

Recently reported analyses indicate that a high proportion of carbon in secondary effluents is in fulvic acids and low molecular weight compounds with detergent and chelating properties. The initial mobility of these materials and of associated metals and non-metals will be high, due in part to high enrichment with N.

At hydrologically feasible rates, the input of organics in the effluent considered will be small (210 lb total C at 35 inches per year). Nevertheless, long term considerations call for management of water and surface vegetation to promote stabilization of mobile components in the soil column in depth, but prior to discharge into drainage or ground water.

Management parameters to be considered include: (1) selection and management of crops for maximum production and return of unharvested residues; (2) fluctuating water regimes and rotational sequences to promote distribution of mobile compounds in depth under vegetation tolerant to excess water, followed by deep rooted crops and reduced water input to promote dewatering and stabilization; (3) use of lime to maintain optimum pH for maximum production of crops and to promote condensation reactions leading to stabilization; (4) minimum tillage to minimize disruption of structural arrangements which, in undisturbed soils, reduce the accessibility of stabilized organics to continuing microbial attack.

At 35 inches per year, the net increment of mineral N (input plus N released by mineralization) was estimated at 46 to 145

lb/acre/year for two different effluent profiles. Neither of these values are equivalent to fertilizer N requirements for yields of corn or forage crops attainable with irrigation on the soils of the projected acceptance areas.

At 70 inches per year, the mineral N increment of 291 lb/acre/year for the effluent containing more total N, more extensively mineralized and nitrified, would support near maximum yields of corn and forage crops. However, hydraulic conductivity of subsoils would very likely be inadequate to transmit water at rates necessary to maintain aeration conditions favorable for legumes, brome grass or corn at this irrigation rate.

Timothy and reed canarygrass might maintain high productivity at 70 inches/year if irrigation on unfrozen soil could be extended into winter and early spring when these species are dormant and can withstand prolonged waterlogging and periodic flooding.

The proposed cropping system lends itself to rotational cycles of wetting and drying which may be manipulated to promote long term improvements in subsoil permeability.

The high proportion of grasses in the rotation and the high moisture regimes projected lead to predictions that immobilization in accumulating residual humus will promote significant inactivation of potentially toxic industrial metals and non-metals and, in conjunction with denitrification, will lead to negligible movement of nitrogen into drainage at irrigation rates which do not exceed soil hydraulic performance appropriate for high yields of all crops in the rotation.

#### Heavy Metals

The relatively high clay content of the soils proposed for the effluent waste disposal site will provide a high degree of adsorbing capacity for most trace metals.

Arsenic will compete for fixation sites with phosphate but will be less tightly bound. If the phosphate fixing capacity of the soil nears capacity, there will be some movement of arsenic

through the profile and possibly into the drainage water.

If boron is applied at the projected rates, there will be movement through the profile into the drainage water and there may be danger of plant toxicity depending upon the tolerance of the crop grown.

Cadmium, lead and mercury are being added at sufficiently low levels that they will be fixed by the soil profile and probably in the top few inches. They pose no plant toxicity danger unless mercury is converted to an organic form such as methyl mercury.

Cobalt, copper, manganese and zinc are added at higher rates but should be fixed by the soil exchange capacity due to the relatively low rate of annual addition. They should pose no danger to plants or to drainage water contamination under the proposed conditions of the system.

The soil should contain adequate fixing capacity for chromium to prevent leaching to drainage water; there is no danger from plant uptake and only a low possibility of plant toxicity.

Iron and manganese present a different consideration. There is no direct danger of plant toxicity from either of these provided soil pH is maintained at about 7.0. However, the high amount of iron being added may reduce the plant uptake of manganese, copper and zinc as well as other nutrients. The reduced uptake could be great enough in the case of manganese to cause manganese deficiency in certain plants. Also, the iron will compete for adsorption sites with other cations and may cause some metals to move further into the soil profile than would normally be expected.

The effects of projected additions of selenium in the system are not well defined, but the permissible leaching, plant uptake and toxicity levels probably will not be exceeded.

With the exception of boron, possibly arsenic and perhaps selenium there should be little effect on plant uptake and plant growth. The drainage water should meet drinking water standards with the exception of boron and possibly arsenic, iron and selenium.



### Physical Considerations

From the physical data and irrigation-drainage information, these soils can probably infiltrate and transmit two inches of wastewater per week except during rainy periods and during planting and harvest time. If these soils were forced to take 70 inches of wastewater a year severe problems with soil structure and in turn infiltration rates, permeability, soil aeration, redox potentials would all be affected. These would all reduce crop yields. A rate of 35 or 40 inches per year with (1) application early before the crop is planted, (2) an interlude of no or very light application depending on the rainfall during tillage, planting, emergence and establishment of the crop, (3) a period of 2" to 3" application during the three summer months when evapotranspiration is highest, (4) a low or no application period to allow for harvest and then (5) an application period post harvest and before the ground freezes, would avoid most of the problems and provide for near maximum yields of a crop like corn. The greatest hazard in land disposal of wastewater is overloading. Seventy inches per year on these soils would overload them hydraulically. The soils used at Pennsylvania State University (Parizek, R. R., et al., 1967) where two inches per week are applied are more permeable than those in this study and this fact must not be overlooked.

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